

November 1984

RESEARCH AND DEMONSTRATION OF IMPROVED METHODS FOR
CARRYING OUT BENEFIT-COST ANALYSES OF INDIVIDUAL

by

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VOLUME II

BENEFIT METHODOLOGIES APPLIED
TO ECOLOGICAL HAZARDS FROM TOXIC SUBSTANCES

Final Report
Submitted to U.S. Environmental Protection Agency
Cooperative Agreement 68-809-702-01-0

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PART 4

METHODS FOR ESTIMATING THE BENEFITS FROM MITIGATING ECOLOGICAL DAMAGES FROM TOXIC CHEMICALS

Robert Repetto

I. INTRODUCTION

Most of the concern and even more of the regulatory action over toxic chemicals in the environment have been motivated by their threats to human health, especially their potential carcinogenicity. This cancer focus has emerged despite the response to early threats by Rachel Carson (Carson) and others that non-human species were undergoing harm. More progress has been made in understanding the effects on natural populations. Yet, there are substantial values that are at risk in possible injury to plant and non-human species.

There are various philosophical positions from which these values might be derived. As discussed below, these include utilitarian and non-utilitarian perspectives. At the simplest utilitarian level, there are various kinds of direct commercial loss. Chemicals have resulted in massive fish kills and closure of commercial fisheries. Studies of the indirect costs of pesticides have identified substantial damages due to honeybee poisoning of farm and household animals: crop losses due to pesticide drift or persistence in the soil after the cropping period of application; harm to beneficial insects and soil organisms (Pimentel et al.). Reasonable estimates of these

commercially valuable losses are in excess of a billion dollars per year in the United States.

In addition, quite aside from commercial and other directly productive benefits, millions of Americans care significantly about and actively enjoy natural populations. According to a 1980 survey, over eighty-three million adult Americans in that year participated in recreational activities primarily motivated by the desire to enjoy wildlife in nondestructive ways (U.S. Department of Interior). Almost twenty-nine million adults took trips primarily to enjoy wildlife, involving 377 million person-days. Five billion person-days were spent in activities around the home primarily oriented toward the enjoyment of wildlife. For example, more than sixty-two million adult Americans, almost one in three, feed wild birds. In short, aside from any commercial or instrumental usefulness, there is extremely widespread interest in natural populations for their own sake.

The corresponding expenditures are equally impressive. In 1980, an estimated \$6.6 billion was spent on activities primarily oriented to the enjoyment of wildlife, including \$2.6 billion on equipment like birdseed and bird feeders, film and photographic gear; and \$4.1 billion on direct travel expenses.

Other recreational activities dependent on natural populations, such as hunting and fishing, are equally popular. In 1980 42 million adults went fishing, spending a total of 858 million person-days in 750 million fishing trips, and spending \$17.3 billion in direct costs. These figures are

indicative of the substantial values, utilitarian and non-utilitarian, that might be endangered by damages caused by toxic effects of chemicals in the environment.

The order of magnitude of these values is easier to establish than the severity of damages toxic chemicals have caused to natural populations. Although much is known about the toxicity of specific chemicals which have been subjected to testing to animal and plant species, much less is known about the effects of the same chemicals on the corresponding natural populations or to populations of species not subjected to testing (Brown). This is due in part to the complexity of ecological systems. The symptoms of damage to an ecosystem often appear far from the biological point of entry of the toxic chemical into the system. Thus, reductions in the population of fish-eating birds and their avian predators due to chlorinated organic pesticides from eggshell thinning is an example of damage appearing far up the foodchain from the biological point of entry of the pesticides. Empirical ecological models are often unable to predict population responses to such chemical intrusions.

Further, risk assessments of potentially toxic chemicals carried out as part of the regulatory process rely heavily on laboratory tests of acute and chronic toxicity to a small number of test species, which are chosen for their ubiquity and ease of rearing in captivity (U.S. E.P.A.). Very limited testing of effects at the population or community level is carried out. Nor is there adequate methodology or capability at the command of the regulatory agencies to monitor the impacts of chemicals in use on

natural populations. For these reasons it is extremely difficult to assess the implications of laboratory test results. There are numerous examples of discrepancy between laboratory and field results. A chemical may have no toxic effects on a particular species in the lab, but still result in significant reductions in field populations because of modification for other environmental factors in the field, because of toxic effects on food supply, or perhaps because the chemical induces behavioral changes that increase losses to predation. A chemical may adversely affect reproductive success or mortality of a species in lab tests, and yet have no discernible effects on natural populations, or even positive effects due to greater reductions in predation pressure (Luck et al.). These complex effects are not well illuminated by methods of risk assessment and toxicity testing in current use.

Finally, of the thousands of potentially toxic chemicals in commercial use that find their ways into the environment, only a small number have ever been subjected to rigorous scrutiny and testing for ecological risks. Many were initially marketed before current regulations calling for such testing were in force. The process of reexamining chemicals in current use is time-consuming. New tests, some of which take years to perform and evaluate, are required. Information on production volumes, use patterns, disposal routes, environmental fate, and exposure patterns, must be collected and analyzed. Regulatory procedures require detailed justification of regulatory decisions and attention to contrary evidence in public documents (National Research Council). Therefore, reexamination may take several years to complete. Under TOSCA, risk assessment efforts are

concentrated mainly on new chemicals and small numbers of priority substances among existing chemicals. In the pesticide field, of the 600-odd active ingredients in over 35,000 registered products, less than ten percent have completed the reregistration examination required by FIFRA and FEPCA (U.S. E.P.A., 1982). The large majority of potentially toxic substances to natural populations have not been subjected to rigorous testing or risk assessment. This adds to the uncertainties that result from the limitations of prevalent risk assessment methodologies and test systems.

Therefore, apart from effects on a few species of falcons, hawks and eagles, whose populations have been adversely affected by pesticide bioaccumulation (along with habitat loss and other changes), little can be said with confidence about the long-term effects of toxic chemicals in the environment on natural populations. Illustrative and interesting is a study of shift in the abundance of common bird species in England over the extraordinarily long period between the 1830's and the 1960's, based on observations of British birdwatchers and ornithologists. In the period after 1939-45, when chemical pesticides came into widespread use, sixty bird species apparently increased in abundance, while thirty-seven declined. However, in the hundred years prior to World War II, forty-one species increased, while forty-seven declined. This ambiguous finding is illustrative of the uncertainty surrounding long-term effects on populations.

Recommendations regarding improvement of the assessment of ecological risks from chemicals have emphasized (a) the need for more sophisticated use of ecological modeling, (b) more testing for effects at the population and community level, and (c) greater efforts at field monitoring of chemical impacts.

II. ECONOMIC EVALUATIONS OF TOXIC EFFECTS ON NATURAL POPULATIONS

Given these uncertainties in the assessment of physical effects and risks to natural populations from toxic chemicals in the environment, it is reasonable to ask what priority should be assigned to improvements in the economic valuation of those risks. Attempts to assign monetary values to ecological risks are debatable in principle and tentative in execution. Would not such attempts merely compound uncertainty, adding little or nothing to the decision process?

The answer is no, for several reasons. The quantification of potential economic damages from toxic chemicals in the environment is useful, first of all, because it forces explicit consideration of the losses that are threatened, their nature, incidence and magnitude. The quantification effort, within an appropriate analytical framework, provides guidance for data collection, and may shed light on the most suitable design for physical risk assessment. This is demonstrated below, in the context of pollution damages to fisheries or other harvested biological stocks. Secondly, approximate estimates of potential economic losses may be useful in deciding on priorities among chemicals for future detailed risk assessment. Current procedures used in EPA's program offices for toxic chemicals in assigning priorities to chemicals for future regulatory examination have moved in this direction by employing indices of toxicity and exposure to suggest total risk, without, however,

including indices of relative economic importance. Thirdly, appropriate estimates of potential economic losses can shed light on the value of additional testing and research. Cost and time considerations have favored heavy reliance on short-term, relatively inexpensive toxicity tests, as opposed to field monitoring and testing at the population or community level. Whether more elaborate testing of ecological risk is worthwhile depends heavily on the range of potential damages that might be incurred through incorrect regulatory decisions. Finally, the experience of most practitioners with benefit: cost analysis is that the procedure is helpful to decision-makers by uncovering gross imbalances between marginal benefits and costs attendant on a regulatory change, often by examining benefits (or costs) that had previously not been estimated at all. Oftentimes, unexamined values can be severely under- or overestimated. Benefit: cost estimates are not devices for fine tuning, but gross checks on the wisdom of regulatory decisions.

The Basis For Valuation

However desirable quantification of potential economic losses to natural populations from toxic chemicals might be, the practical methodology for making such estimates is weak, and the philosophical foundation for estimation is contested. Many people reject the economist's utilitarian assumption if applied to the preservation of natural environments (Leopold). These objections often have religious grounds: Natural environments should be protected because they are God's creations and sacred. Mankind has been given stewardship over creation, with a

responsibility for its protection. Objections, if not explicitly religious, may reject utilitarianism as hopelessly anthropocentric, denying both our "kinship" to other living beings and their "rights" to exist and thrive (Favre, Stone).

These positions can be debated, accepted or rejected, but never proven or disproven, because they are assertions about the way society ought to manage itself.

Nonutilitarian positions have substantial acceptance in the political arena, as demonstrated by passage of the Endangered Species Act, which does not require economic justification for species preservation (Endangered Species Act). Yet, there is continual encroachment on natural environments and populations as the result of economic activity without serious political repercussion. Consistency cannot be found in the political record. Personally, most of us draw uneven and fuzzy boundaries for our own behavior. Clear and consistent consensus about our proper societal obligations to natural environments is unlikely to emerge (Kellert).

Even were there consensus around a nonutilitarian basis for valuing natural populations, it would be difficult to base social decisions on it because such bases are typically absolute, and do not readily admit trade-offs. If other species have rights, how are these rights bounded and balanced? How could nonutilitarian values be derived to compare with the utilitarian opportunity costs of human welfare? For the most part, these issues are not addressed by those who assert the inadequacy of a utilitarian foundation.

One proposal has been advanced to find a common denominator for economic and ecological activity: that both should be valued in terms of our system (Odum). Ecological systems have been modelled as akin to competitive economies with individual plants and animals in atomistic competition seeking to maximize net energy gain (energy intake less expenditure) (Hannon, 1978, 1979; Rapport and Turner, 1977). Similarly, "energy theories of value" have been advanced arguing that economically produced goods and services should, and largely do, exchange in proportion to their total energy content. Adoption of such a framework would indeed lead to a calculus wherein the ecological and economic effects of the release of toxic substances into natural environments could be related, at least in principle: the loss in stored energy in the ecosystem against the energy costs of abatement of the emission. The shortcomings of this principle, both as normative and as positive valuation method, are obvious. It is a poor predictor of relative market prices in the economy. Nate "Tiny" Archibald of the Boston Celtics and I are about the same size and both play basketball. It is doubtful whether the difference between his market value as a basketball player (approximately \$300,000 per year) and mine (approximately nothing) can be explained in terms of relative energy outlay. The energy theory of value is also a poor normative system for valuing changes in natural resource stocks. Valuing sperm whales at their BTU value would hardly be considered a step forward in the methodology of environmental economics.

At least for purposes of this analysis, we are thrown back on the economist's utilitarian assumption that values are based on, and to be derived from, individual preferences. The natural environment is worth what it is worth to us, for whatever set of reasons. This is not as restrictive as it may seem, for, while it does not accept the position that social welfare ought to be denied as dependent directly on the natural environment, it admits as valid the preferences of those individuals who believe so, and their possibly very high valuations of aspects of nature. These distinctions can be clarified by distinguishing three possible versions of the social optimization problem, formulated in terms of a general environmental good E and an aggregate final consumption good X.

Position 1. The Natural Environment Only as Input

Natural populations are valued only instrumentally for their life-supporting functions, because they permit and increase the production of goods and services that do not enter final production flows. Thus, the genetic resources of rare species are valued for their possible use in plant breeding. Insect populations are valued as predators on other insect pests. Marshlands are valued as spawning grounds, habitat, flood and pollution control devices, and so on (Myers, DeBach). As depicted in Figure 1, the aggregate production functions $f(X,E) = 0$ attains a maximum level of X for some positive E, reflecting both the life-supporting function of the natural environment and

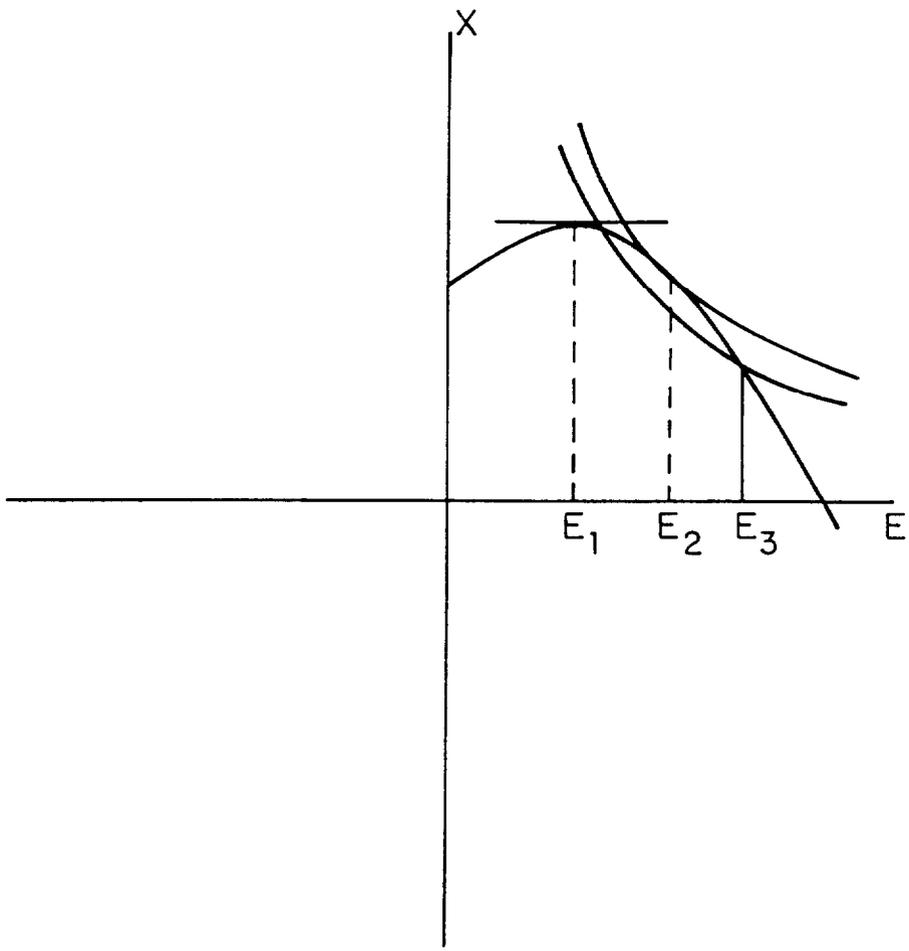


Figure 1. Aggregate Production Function

the competition between economic and natural production for primary resources. Since, according to this position, only economic outputs are directly valued, the social welfare maximization problem for N consumers is

$$W = W[U^1(X_1), \dots, U^n(X_n)] = \max. \text{ subject to} \quad (1)$$

$$X_1 + \dots + X_n = X \quad \text{and} \quad f(X, E) = 0.$$

The environment is worth preserving only up to the point E_1 in Figure 1 where $dX/dE = 0$.

Position 2. The Natural Environment as Utility-Yielding

Not only does nature support economic production, it provides direct satisfactions to those who contemplate it. The social welfare maximization problem is

$$W = W[U^1(X_1, E), \dots, U^n(X_n, E)] = \max. \text{ subject to} \quad (2)$$

$$X_1 + \dots + X_n = X \quad \text{and} \quad f(X, E) = 0.$$

The environment is worth protecting to the point at which the trade-off between natural and economic production equals the relative valuation of the two, where, because the natural environment is a public good, the relevant valuation is the sum of the individual marginal rates of substitution. The point E_2 in Figure 1 corresponds to the point at which

$$U_X^1 / U_E^1 + \dots + U_X^n / U_E^n = dE / dX \quad (3)$$

This is the standard utilitarian formulation, which gives rise to the problem of estimating aggregate willingness-to-pay for environmental quality.

Position 3: The Natural Environment as Non-Utilitarian Good

This position asserts that, apart from individual preferences or its life-supporting functions, the natural environment is worth preserving for its intrinsic value. This position, in the language of the literature, makes environmental quality a collective "merit" good. One way of formulating this position is to assume a minimum level of environmental quality that is socially acceptable. The social welfare problem then becomes

$$\begin{aligned} W = W[U^1(X_1, E), \dots, U^n(X_n, E)] = \max. \text{ subject to} & \quad (4) \\ X_1 + \dots + X_n = X \text{ and } f(X, E) = 0 \text{ and the additional} & \\ \text{constraint } E \geq E_{\min}. & \end{aligned}$$

This formulation may lead to a position such as that corresponding to E_3 , where E_3 corresponds to the minimum acceptable level of E .

Within the utilitarian framework, the potential Pareto-improvement criterion for evaluation of a proposed improvement in environmental quality, regarded as public and indivisible, requires that the Hicksian compensating variation concept of consumer surplus be employed as the measure of individual willingness-to-pay, and the compensating willingness-to-accept be used for evaluation of decrements to environmental quality (Bradford, Brookshire et al., Mishan). The value of changes are assessed with reference to the initial welfare position of those affected. In terms of the individual bid curves depicted in

Figure 2, with X and E again representing the consumption commodity and the environmental good respectively, compensating and equivalent variations are indicated for equal increments and decrements in the level of the environmental good, with the origin taken as the initial position. (Willig, Randall and Stoll). It is known that for equal changes,

$$WTA^C = WTA^{Eq} > M > WTP^{Eq} = WTP^C, \quad (5)$$

where C and Eq represent compensating and equivalent measures and M represents the Marshallian consumer surplus (the area under the uncompensated demand curve for E). For discrete changes in environmental quality, the approximate limits on differences between willingness to pay and willingness to accept have been established:

$$WTA - WTP \approx aM^2/X, \quad (6)$$

where a is the price flexibility of income and aM is small relative to X. This finding has reassured applied economists that errors involved in using Marshallian or either of the Hicksian concepts of consumer surplus in empirical work in accordance with feasibility or convenience would generally be small relative to other uncertainties in the problem. However, numerous explorations of willingness-to-pay through survey techniques and direct observation have unearthed a difficulty. Differences between WTA and WTP are usually substantially larger than can be explained through income effects (Schulz et al.; Gordon and Knetsch; Bockstael and McConnell). Differences are often one or more order of magnitude, large enough potentially to

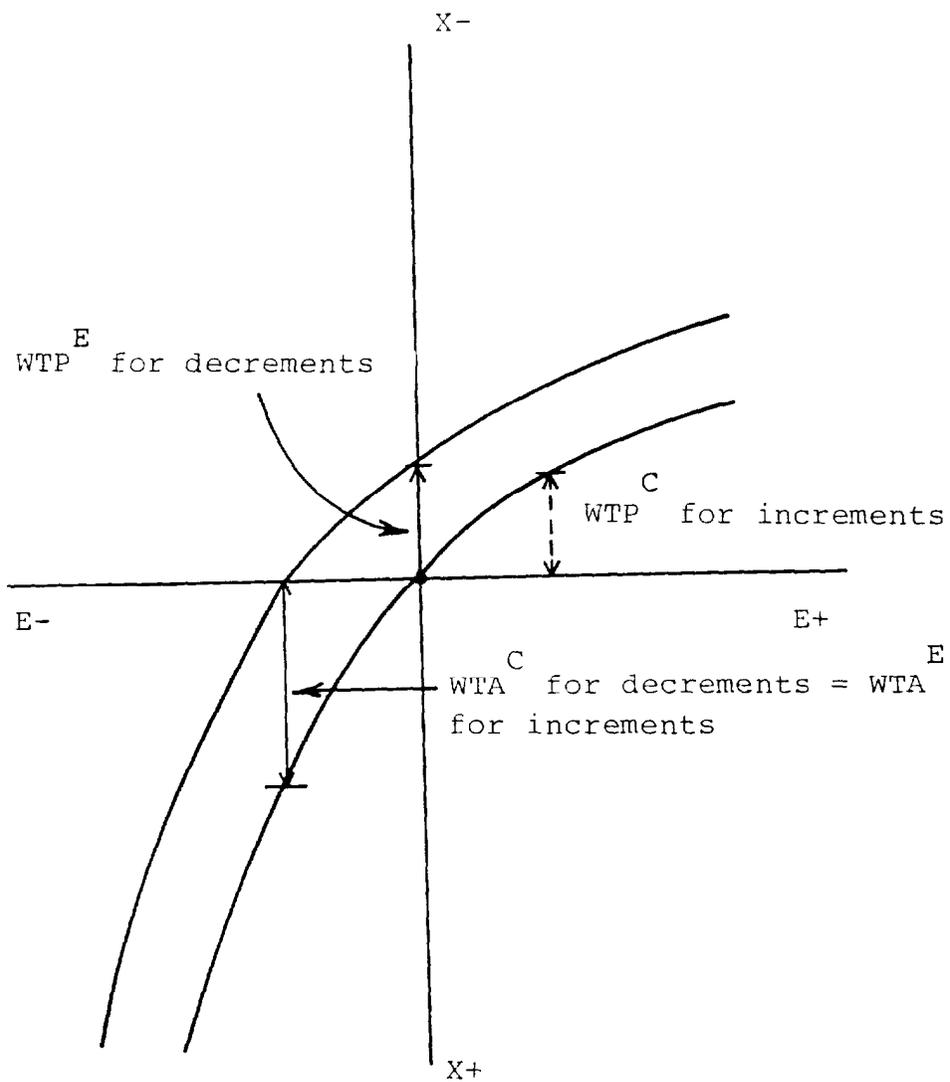


Figure 2. Individual Bid Curves

affect the conclusions of decision-oriented studies.

It has been suggested that WTP concepts should be preferred for empirical estimation over WTA measures, because the former are less likely to be distorted because they are subject to budgetary constraints. It has also been suggested that compensating measures, for which the initial and the reference levels of welfare coincide, are likely to yield more reliable estimates (Brookshire et al). However, there is evidence that the discrepancy between WTA and WTP values observed in empirical research reflects systematic and regular features of individual behavior. These features of behavior apparently contradict the predictions of economic theory with respect to consumer behavior, especially in choices involving uncertainty. Attempts have been made to describe these behavioral features systematically in what is called "prospect theory." Two aspects relevant to our problem are that

- first, if out-of-pocket costs are viewed as losses and opportunity costs are viewed as foregone gains, the former will be more heavily weighted, so that individuals will be more unwilling to pay than to forego compensation:
- second, if a good is viewed as already part of the individual's endowment, implying an assumed entitlement to it, it will be more highly valued than a good not already in the individual's endowment. This implies that compensation to accept reductions in existing environmental amenities will have to be greater than willingness to pay for comparable improvements.

These two features of individual psychology, which seem to be empirically demonstrable, imply that bid curves such as those drawn in Figure 2 should in fact be drawn kinked at the origin, with losses in environmental quality valued more heavily than gains, as indicated in Figure 3. It also suggests that attempts to estimate WTP^E for decrements in environmental quality may encounter difficulties, because the concept involves out-of-pocket costs and the potential loss of welfare from the individual's initial endowment. The difficulty that prospect theory, and the regularities of behavior on which it is based, create for the valuation of environmental amenities is that it predicts that there will be large systematic differences between WTA and WTP which cannot be explained through income effects and cannot be explained away as aberrations of specific studies. The implication seems to be that the proper measure must be selected in view of the actual distribution of initial endowments of the specific change under investigation, whether environmentally favorable or unfavorable. If, in the consumer's subjective estimation, given physical changes are valued very differently depending on whether they are regarded as losses or gains, and whether out-of-pocket costs to the consumer are implied or not, then the estimation of benefits will be more accurate if the appropriate measure of consumer surplus for the situation at hand is adopted.

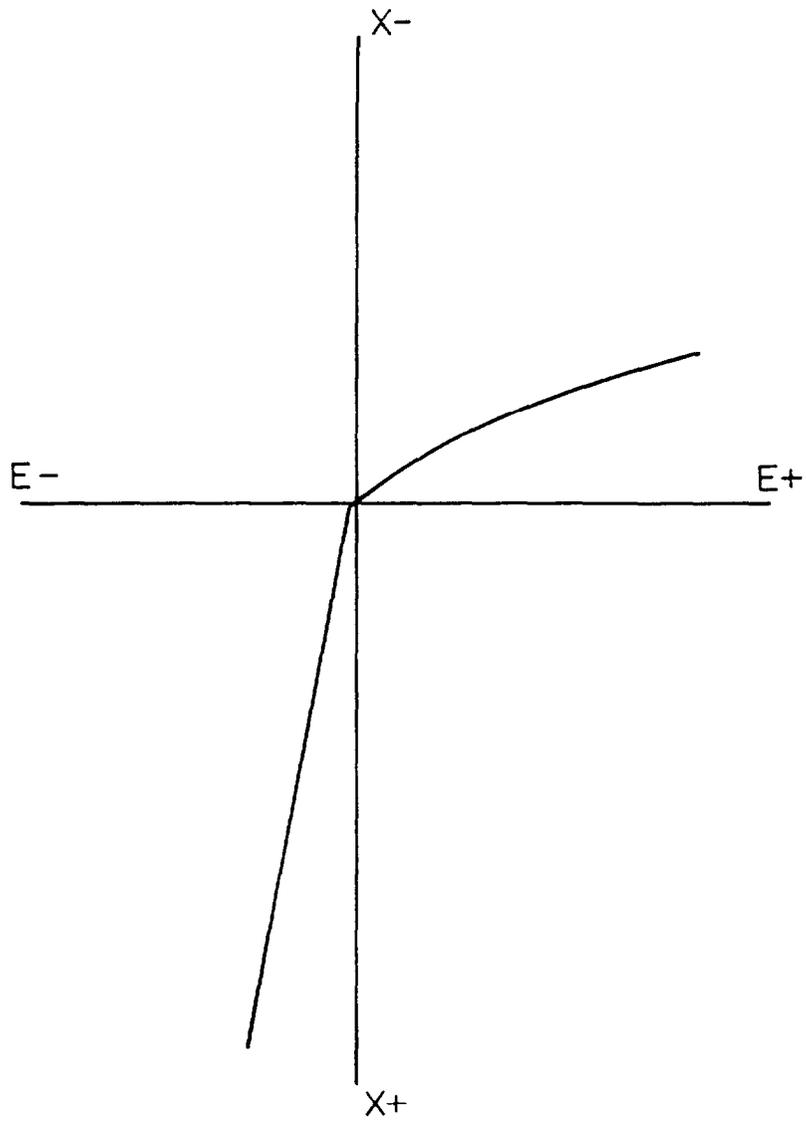


Figure 3. Revised Bid Curve

III. METHODOLOGIES FOR EVALUATING TOXIC EFFECTS

It is crucial to understand that the relevant task is to estimate the damages done by toxic chemicals to natural environments. Toxic chemicals may be present in air, land or water bodies in concentrations of parts per million or even billion, and be impossible to detect without sophisticated measurement equipment and techniques. Unlike, say, a smokestack plume across a scenic vista, it is not their presence which is itself objectionable, but their effects on plant and animal communities. It is the value of these effects that can be combined with values assigned to health effects and compared with abatement costs.

The earlier discussion of risk assessment methodologies emphasized the importance of considering the population dynamics of affected species, in order to understand the population significance of physiological impacts. From the standpoint of economic evaluation, population dynamics are equally important, because they call attention to what is, in effect, the production function for natural populations. Toxic chemicals may be considered as negative inputs into production functions for species populations, and operate along with other inputs: habitat availability, climatic variables, and so on. In terms of population dynamics, these production functions can be written generally as

$$\dot{E} = f(E;A,M), \text{ with } E \text{ and } f(\) \text{ vectors} \quad (7)$$

for multispecies models, with A representing ambient concentrations of toxic chemicals and M representing other variables influencing the growth of natural populations. If steady-state solutions exist, there is a corresponding static relationship

$$E = g(A,M). \tag{8}$$

Therefore, even though, under the utilitarian approach adopted above, the component stocks of the natural environment enter directly into individual utility functions, toxic chemicals can be regarded as intermediate inputs, for which the valuation is derived from final demands. The distinction is important, because estimates of the (negative) value of toxic concentrations in the environment as factors of production can often be derived from market data without addressing directly the difficult issues of consumer willingness-to-pay.

IV. CONSUMPTIVE USES AND THE DYNAMICS OF NATURAL POPULATIONS

The idea of a biological production function underlying the dynamics of natural populations is central to the evaluation of pollution damages to harvested species, because such damages interact with commercial or recreational removals of the stocks in complex ways. At one extreme, mortality from toxic substances to species already subject to heavy harvesting pressure might tip the dynamic path of the population irreversibly toward extinction. At the other extreme, reduction in demand or increased harvesting costs due to pollution effects might allow an over-exploited stock to rebuild toward more optimal levels, increasing long-run welfare.

In general, it will be desirable to distinguish whether toxic substances affect the biological production function of the population in question, or the economic behavior of harvesters or consumers directly; if the former, whether the habitat and carrying capacity of the species are affected or fertility and mortality rates; and, if the latter, whether the demand for the harvest is affected, or the costs of harvesting. It will also be important to identify the institutional structure within which the species is managed. Welfare effects of pollution in an open-access fishery may differ substantially from the effect if the same fishery were optimally managed (Clark, Levhari et al.).

This section illustrates consumptive use, which may be thought of as a commercial fishery for the sake of concreteness. An approach is outlined for valuing pollution impacts of various kinds, and a possible empirical application is indicated.

Only single-species models of the form $E = f(E;A,M)$ are considered. Multi-species models cannot easily be treated analytically. The main distinction in this growth equation, between production processes which display pure compensation and those that display depensation is illustrated at the top row of Figure 4. As illustrated there, compensation models are those for which $f(E)/E$ declines monotonically. For depensation models, $f(E)/E$ increases over some range of E ; and, for processes displaying critical depensation, $f(E)$ is negative for some range. These last are important, because if critical depensation is present at $f(0)$, even intermittent pollution losses can lead to the extinction of the population without any harvesting activities. The best known model is the logistics, for which

$$\dot{E} = [r(1-E/K)]E \quad (9)$$

The maximum growth occurs when $E = K/2$, and the stock E approaches K asymptotically. If the stock is harvested at a constant rate H , then the growth equation must be modified.

to $\dot{E} = f(E;A,M) - H$. The maximum sustainable harvest, in the logistics model, is clearly attained when the stock is $K/2$ and the maximum sustainable harvest is therefore $rK/4$.

The link between the biological and economic processes is through harvesting behavior. The harvest depends on a measure of

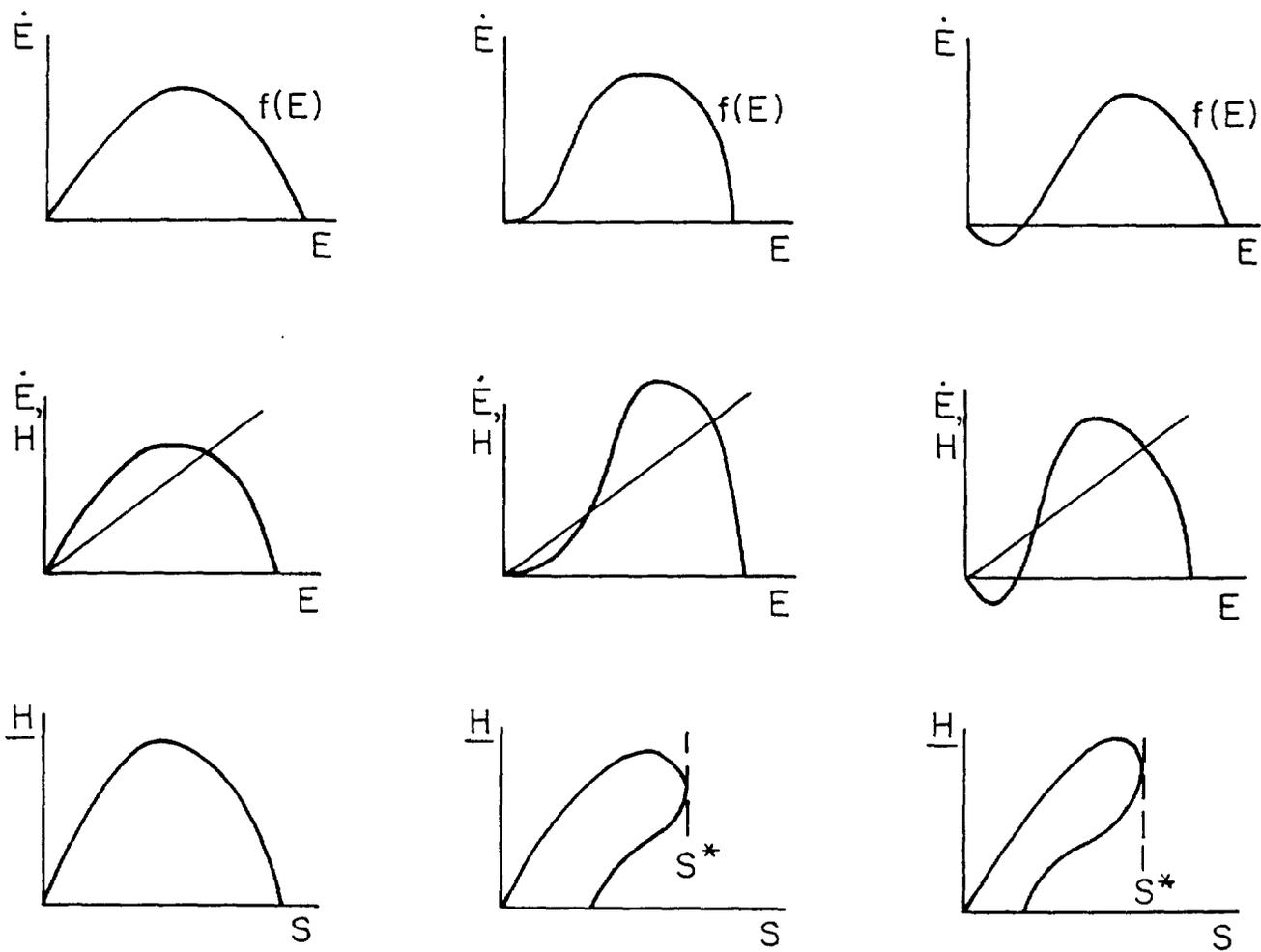


Figure 4. Biological Production Functions, Harvest Functions, and Yield/Effort Functions

harvesting effort, S , which should have the dimension fishing lines (nets, boats, or traps) per day, as well as the size of the natural stock. The simplest version of this economic production function specifies $H = (qS)E$, or simply $H = SE$, if the units of S are redefined to make q , the productivity of a unit of effort, equal unity. The bioeconomic production function, $E = f(E;A,M) - SE$ then becomes, in the case of the logistics example,

$$\dot{E} = [r(1-E/K) - S] E. \quad (10)$$

In the second row of Figure 4, the harvesting functions are superimposed as rays on the biological growth equations. For compensation models, any level of effort up to $f'(0)$, which is the intrinsic growth rate r in the logistics model, results in a unique sustainable harvest level, with a maximum sustainable yield (MSY) at E_{max} . For depensation models, any harvest level might result in two sustainable yield levels, of which one would be unstable. At the latter, any reduction in the stock (by a pollution episode, for example) would lead to further declines in the pollution if harvesting efforts were unchanged.

The final row in Figure 4 illustrates the relationship between sustainable levels, \underline{H} , and levels of effort. This relationship is defined by the equation $f(E) = H$, and, for the logistics process, is

$$\underline{H} = S K (1-S/r) \quad \text{for } S \leq r. \quad (11)$$

For compensation processes, $(dH/dS)/S$ declines monotonically with increasing effort. For depensation processes, however, there is a maximum sustainable level of effort S^* beyond which the maximum sustainable yield is zero. The lower branch of the yield-effort function is unstable.

The bioeconomic model is completed by specification of demand and supply functions for the harvest. The simplest demand assumption, that price is given and invariant over harvest levels, would be appropriate if the fishery affected by the toxic pollution contributed a small fraction of total market supply. The simplest supply assumption, that the marginal costs of increasing levels of effort are constant, would be appropriate either for short-run analysis, in which capital costs are considered fixed and sunk, or long-run analysis, in which all costs are considered variable. Constant marginal costs of effort, of course, do not imply constant marginal costs of harvest, since these costs will vary inversely with the stock.

Institutional structure is important in the specification of economic behavior. In the open-access fishery, the individual participant does not take into account the effects of his harvest on the harvesting costs of others (Bell). Anyone who has seen a rocky cove filled with the buoys of lobster pots recognizes this crowding externality. Further, the individual participant in an open-access fishery has no incentive to pay heed to the fact that a smaller catch today could mean a larger stock next year,

because that larger stock next year would only attract more fishermen and not necessarily result in a larger catch for the conservative individual. There is also, therefore, a stock externality.

In the open-access fishery, harvesting effort tends to the level at which total revenues, $pH(S)$, equal total costs, cS . For the logistics example, this results in a determinate level of effort:

$$S = r(1-c/pK) \quad \text{for } c/p < K. \quad (12)$$

If the inequality were reversed, no sustainable level of effort would be profitable, and the fishery would not be commercial (at least continuously). Corresponding to this level of effort is a sustainable yield

$$H = r(1-c/pK) (c/p) \quad (13)$$

and an equilibrium stock, which is simply $E = c/p$.

Resources which are not open-access may be managed quite differently. If, either because it were privately controlled or because it were publicly managed for optimal sustainable yield, harvesting were regulated to maximize the discounted present value of the resource, levels of effort would be determined by

$$PV = \int_0^{\infty} e^{-it} (pH - cS) dt \quad \text{subject to } \dot{E} = f(E) - H \quad \text{and} \quad H = ES \quad (14)$$

This can be rewritten as

$$PV = \int_0^{\infty} e^{-it} (p - c/E) (f(E) - E) dt = \int_0^{\infty} e^{-it} (p - c(E)) (f(E) - E) dt \quad (15)$$

which can be analyzed with classical variational techniques. Euler's necessary condition for a maximum results in the fundamental economic relation of optimal resource management:

$$f'(E) - \frac{c'(E)f(E)}{p - c(x)} = i. \quad (16)$$

Some of the implications of this relation can be seen by noting that, if harvesting costs should be independent of the stock, the stock should be managed to keep $f'(E) = i$: the biological return of the stock must equal the return on other economic assets. This underlies the economic explanation of why slow-growing species, for which $f'(0) < i$, may be exploited to extinction (Clark, Plourde).

An equivalent form of this relation is more revealing. Equation 17 says that, for optimal intertemporal management, the change in the rental value of the resource stock from a marginal unit withdrawn from the stock, $d/dE[(p - c(E))f(E)]$, must equal the return on that marginal unit if harvested and the resulting profits invested at the market rate of interest.

$$d/dE[(p - c(E))f(E)] = i(p - c(E)) \quad (17)$$

This formulation also makes clear the difference between the open-access and the optimally-managed resource. Dividing Eq. 17 through by i and letting i approach infinity, it is clear that the open-access fishery is equivalent to optimal management with an infinitely high discount rate: only current profits are considered. As the interest rate approaches zero, Eq. 17 implies

that optimal management is equivalent to maximizing sustainable rent, with no consideration given to current profits. These behavioral differences due to institutional structure are important in the analysis of long-run welfare effects due to impacts of toxic substances on the resource.

Returning to the open-access fishery, let it be assumed that market price depends linearly on the harvest: $p = a_0 - a_1 H$, and consider the effects of toxic substances on the discounted stream of Marshallian consumer surplus, denoted M .

$$M = \int_0^H (p(H) - c) dH = \int_0^H (a_0 - a_1 H - c) dH = 1/2(a_0 - c)H \quad (18)$$

$$W = \int_0^\infty e^{-it} (M) dt = (1/2i)(a_0 - c)H \quad (19)$$

For the sustainable yield under open-access fishery defined for the logistics example in Eq. 13, this means

$$w = (1/2i)(a_0 - c)r(1 - c/pK)(c/p) \quad (20)$$

where discounted consumer surplus is expressed in terms of the economic parameters of demand, supply and capital cost (a_0, c, i) and the parameters of biological production (r, K). Consider the latter first. It could be that toxic substances in the environment might affect primarily K , the maximum survivable population, if it affected the food supply of the species. Insecticides may reduce bird populations not by poisoning the birds, but by killing the insects that the birds eat.

Alternatively, it might affect r either by reducing fertility or increasing mortality rates. These different impacts will not have the same effects on economic welfare, as the following indicate.

$$\delta W / \delta r = k(c/p) (1-c/pK) \quad \text{where } k = (1/2i)(a - c) \quad (21)$$

$$\delta W / \delta K = kr(c/pK)^2 . \quad (22)$$

For both, the effects on welfare of toxic impacts that reduce biological capacity are negative, provided that the resource is commercially viable, but the magnitudes may be quite different, depending on the size of the growth rate and the size of the stock (remembering that $E = c/p$).

It is possible that toxic impacts may affect welfare even though the biological production function is not altered at all. Think of a commercial fishery which must be closed either in part or part of the time because of pollution levels that would exceed health tolerances for human consumption of the fish. Assuming that the chemicals would not be toxic to the natural population, the effect would be an increase in the cost of harvesting. It is not even clear what the direction of the welfare effect of such a toxic impact would be. If the resource were overexploited under open-access harvesting, the welfare effect of an increase in the costs of production might be positive. After all, certain fisheries are regulated by artificial increases in production costs (like handtonging for oysters from powerboats or dredging for oysters only under sail in Chesapeake Bay) or temporary closures. Examination of Equation 23 confirms the ambiguity.

$$\delta W / \delta c = k(r/p)(1-2c/pK) - (r/2i)(1-c/pK)(C/P) \quad (23)$$

This equation is quadratic in the level of costs and could take on positive or negative values. Since the long-run supply curve, which is the relation given in Eq. 13, is backward-bending in the cost-price ratio, so that an increase in costs may either raise or reduce long-run sustainable harvest, the welfare effect of pollution that raised harvesting costs might raise or reduce economic welfare.

Finally, toxic impacts may affect the price obtainable for a given harvest, without affecting biological production or harvesting costs (Swartz and Strand, Shulsted and Stoevener). When kepone contamination forced the closure of the St. James River in Virginia to fishing, especially shellfishing, the demand for oysters and oyster prices fell up and down the entire Atlantic seaboard, even in fisheries far removed from Chesapeake Bay. For reasons like those advanced for cost increases, the welfare effects of a price decline may be positive or negative in the open-access fishery. If prices are already "too" high, the crowding and stock externalities may have resulted in inflated production costs and over-exploitation of the resource, so that a price reduction would increase economic welfare. Equation 24 illustrates this phenomenon for the logistics case.

$$\delta W / \delta a_0 = k(rc/p^2)(c/pK-1) + (r/2i)(1-c/PK)(c/P) \quad (24)$$

It is clear that these results do not hold, even qualitatively, for the optimally-managed fishery. Since crowding and stock externalities are taken into account and harvests

regulated to maximize the discounted present value of the resource, any impairment of biological productivity or increase in harvesting costs or reduction in the economic value of the harvest must result in a reduction of social welfare. The quantitative impacts will still differ, of course, depending on the nature of the adverse impact of toxic substances. Therefore, it is important to identify the institutional structure of the harvesting activity.

There have been attempts to use this framework to estimate the consumer surpluses obtained from consumptive uses of a biological resource, and to estimate the welfare effects of a change in environmental conditions (McConnell, Hammach and Brown, Lynne et al.). The general procedure is to estimate the biological production function and the demand and supply conditions for the harvest, letting environmental conditions affect one or more of those functions. With these estimates, the welfare effects of environmental changes can be calculated. In the example of the logistics process, the biological parameters (r, K) might be estimated from Equation 11. Demand and supply functions would be estimated by conventional methods to yield an empirical version of Eq. 20. Alternatively, if data on a range of cost-price ratios were available, Eq. 13 might be treated as an estimating equation for the biological parameters, or might be modified to include the effects of pollution. The difficulty is that these relations define steady-state equilibria, unless the observations are averaged over considerable periods of time. Moreover, specification errors in the biological production

function can have serious effects on the estimates. It might be preferable to concentrate empirical work on the demand and supply functions for the consumptive use, and rely on exogenous estimates from natural scientists for the biological production function, and the impact of toxic substances on it.

V. TOWARD EMPIRICAL ESTIMATION OF THE IMPACT OF TOXIC
SUBSTANCE CONCENTRATIONS ON NONCONSUMPTIVE ENJOYMENT
OF NATURAL POPULATIONS

The value of consumptive and nonconsumptive recreational uses of the natural environment, as indicated by time and money expenditures, must be substantial. An ability to arrive at -- even approximate -- simple estimates of the impact of toxic damages on these values would be important in improving regulatory decision-making. Such estimates would be helpful in deciding how to assign priorities to chemicals in scheduling detailed regulatory proceedings, given the huge backlog of pending work and the length of time such regulatory analyses consume. The ability to estimate the order of magnitude of potential damages would also be useful in deciding how much testing to require of potential registrants, and what resources to devote to field monitoring programs. The value of additional information is relegated to the potential losses from incorrect regulatory decisions; hence, in part, to potential losses from toxic damages.

Therefore, an estimation methodology based on the notion of averting or offsetting costs by private and public parties is first presented. This approach is based on potential substitution among inputs in the biological or recreational production function. The negative impacts of toxic chemicals may be counterbalanced by additional private or public inputs. Under some conditions, the cost of these additional inputs may be taken as an approximate estimate of the value of the damages done by

the toxic substances. When this is so, the estimation is converted to an "engineering" problem, in which physical input requirements for offsetting activities are specified and their costs ascertained. This is much simpler than the problem of estimating changes in demand for nonmarketed public goods. This approach is introduced first because of its potential ease in time, data, and analytical requirements.

Private Averting Costs

This approach is based on the concept of the household production function (Bockstael and McConnell, 1981), as well as the environmental production function. Assume that the household produces a recreational service flow by combining marketable inputs with the natural environment: fish plus fishing gear and time combine in producing "fishing"; wildlife plus photographic equipment combine in producing "photographing wildlife," examples of a general recreational activity R that generates utility along with consumption of an aggregate commodity X . The stock of wildlife E is also the result of an assumed steady-state production function. Therefore, the private individual's utility maximizing problem is

$$U = U(X,R) = \max. \text{ subject to} \quad (25)$$

$$R = R[C,E, (A,M)] \quad \text{and} \quad x + p_C C = Y.$$

Here C is a private input into the recreational activity, which could be the cost of equipment or the market cost of time, as in the travel cost model (Bauves and Schneider, Gum and Martin).

The variable A represents the ambient concentration of toxic substances in the production function for natural populations, and the units of X are chosen so that its price is unity in the budget constraint, in which Y represents personal income.

The private optimization process can be thought of as a two-step process, in which the individual first chooses C, given E, to minimize the costs of any given level of R, the recreational experience; then, maximizes utility subject to the price of X and cost function for R.

If in the production function for R, $R_{CC} = 0$, so that $R = Q[E(A,M)]C$, then the budget constraint for the consumer is a straight line. This situation would arise, for example, if Q represented some quality indicator of the recreational experience and C represented a measure of quantity, like the number of days or trips. The marginal cost of the recreational experience would be $p / Q[E(A,M)] = \pi_R$.

The second stage of the process can then be viewed as maximization of the utility function $U = U(X,R)$ subject to the budget constraint $X + \pi_R R = Y$. The resulting demand curve for R can be written as

$$R^* = R^*(Y, \pi_R) = R^*[Y, p / Q(E(A,M))]. \quad (26)$$

Changes in the natural environment E as a result of changes in concentrations of toxic substances result in changes in the shadow price of R. This fact can be exploited to derive approximate cost estimates for toxic damages (Porter-Courant).

The indirect utility function can be written in terms of income and prices:

$$V = V(Y, \pi_R) = V[Y, \pi_R(p, Q)]. \quad (27)$$

The effect of changes in toxic concentrations on welfare can be investigated by differentiating the indirect utility function:

$$dV/dA = V_Y dY/dA + V_{\pi} d\pi/dA. \quad (28)$$

For compensating changes in personal income that would lead to unchanged levels of utility, which would correspond to WTP^C in the previous discussion,

$$dY/dA = -(V_{\pi}/V_Y) d\pi/dA. \quad (29)$$

But, by Roy's Law, $-(V_{\pi}/V_Y) = R$, so that, at the margin, the willingness to pay for improvements in ambient concentrations can be estimated by the reduced cost of maintaining the recreational experience:

$$dY/dA = (R) d\pi/dA = (R) (d\pi/dE) E_A. \quad (30)$$

Similarly, the willingness to accept compensation for deterioration in ambient concentrations can be estimated by the increased cost of maintaining the recreational experience. The situation is illustrated in Figure 5, for the compensated demand curve for R. Improvements in ambient quality lower the shadow price, π_R from π_0 to π_1 , leading to welfare gains that are approximated by the cost reduction in the shaded area.

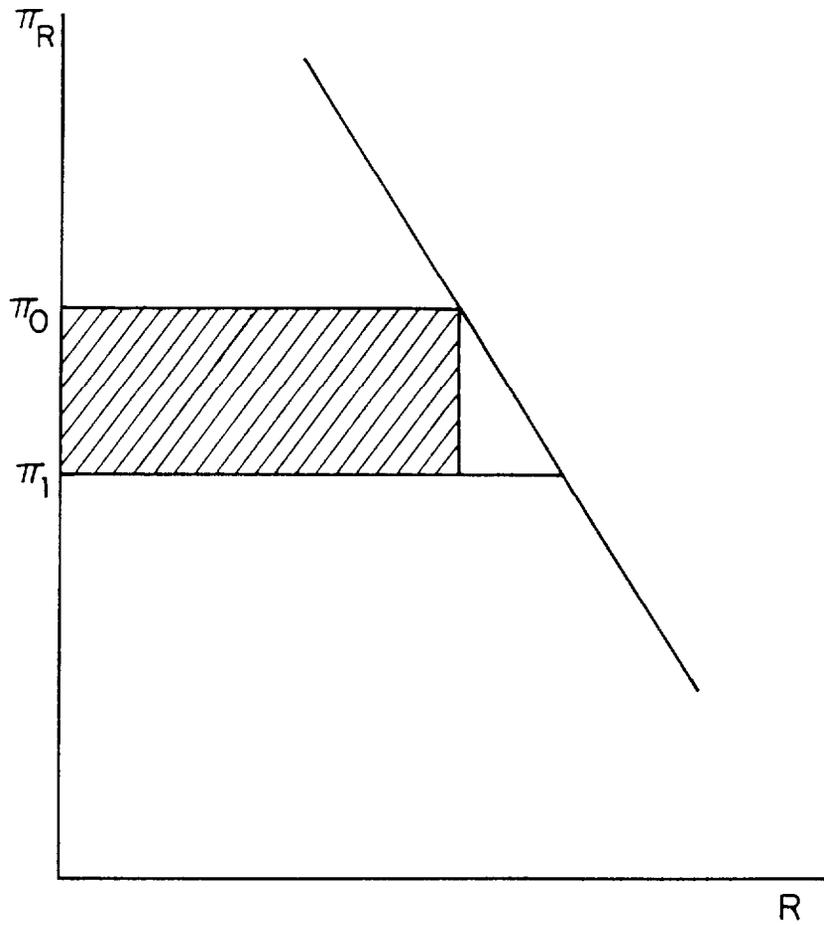


Figure 5. Effect of Improvements in Ambient Quality on Π_R

Applications of this approach arise naturally in the travel cost model. If pollution results in the closure of particular fishing grounds or shellfishing areas, so that recreational fishermen must travel further to go fishing, then the damages can be legitimately estimated as the additional travel costs involved. For example, heavy concentrations of PCB's and other pollutants have resulted in the closure of shellfishing grounds in the upper reaches of New Bedford harbor, Narragansett Bay, and other estuaries. Recreational clammers, lobstermen, and mussel gatherers must travel further to downstream reaches of the outer harbor to gather shellfish. A justifiable and reasonable estimate of their willingness to pay for ambient water quality improvements that would result in the opening of grounds in the upper estuaries is their saving in travel costs and time. The advantage of estimates based on such cost differentials, obviously, is that they are much easier to calculate than are direct measures of the marginal utility of the recreational experience.

This model leads to an alternative route to the estimation of willingness-to-pay, based on the assumption of weak complementarity. The demand for C, the marketable private input into the production of R, is derived from the demand for R. In the foregoing model,

$$C^* = R^*/Q = \frac{R[Y, \pi_R(p_C, Q(E))]}{Q(E)} \quad (31)$$

Changes in environmental quality due to changes in toxic

concentrations result in shifts in the derived demand curve for C, assumed to be supplied at the constant price p_C . The weak complementarity assumption implies, for example, that the marginal utility of changes in the relevant dimensions of environmental quality are zero for those who spend no time recreating. Under this assumption, as illustrated in Figure 6, the consumer surplus attributable to an improvement in quality from E_0 to E_1 can be measured by the area between the two demand curves $C(E_0)$ and $C(E_1)$ (Freeman, Maler). This approach is discussed at greater length below.

Public Offsetting Costs

At this point it is preferable to return to the idea of estimating damages from toxic effects by the costs of offsetting or averting behavior. The approach can be extended to include not only private behavior but also offsetting expenditures undertaken by public agencies in the course of wildlife or ecosystem management. If species or ecosystems are managed in the public interest, and the assumption can be made that management decisions and program expenditures represent a public consensus as implemented by an accountable agency of the optimal state of the ecosystem, then, a feasible methodology for evaluation for marginal damages is calculation of the additional management costs which would be required to restore the natural environment to its prior condition. For example, if a toxic chemical spill reduced the fish population in a particular river (assuming non-persistence, for simplicity's sake) the public costs of restoring that population through increased stocking or

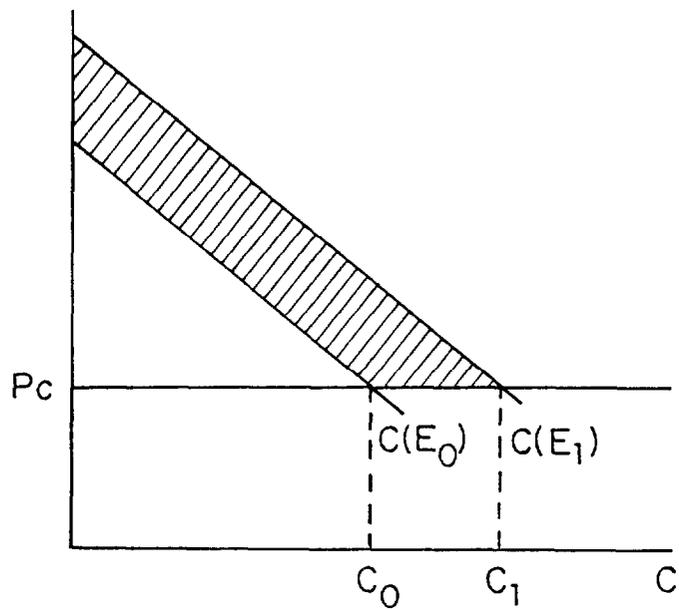


Figure 6. Changes in Environmental Quality Due to Changes in Toxic Concentrations

more cost-effective management alternatives would be a valid measure of damages. If pesticide application resulted in a lowered reproductive success among certain bird species, and those bird populations could be maintained through provision of additional breeding grounds or habitat, the costs of those grounds would be a measure of damages. At the least, the cost of the most cost-effective management program to restore the natural population or ecosystem would be an upper bound to the level of damages.

This can be demonstrated within the framework of the previous model. Let the variable M be interpreted as a collectively determined public management input, such as hatchery output or habitat maintenance. Let it be defined in units such that its price is unity, and financed by lump-sum taxes on personal income. (In fact, such public expenditures are often financed out of taxes and fees on outdoor recreation, which introduces additional complications.) To further simplify the analysis, redefine units of C to make p identical to 1, and let it be assumed that the public agency seeks to optimize a welfare function based on the utilities of N identical individuals. Its problem is then to maximize

$$Z/N = U[X^*(Y,M,A),R(C^*(Y,M,A))] \text{ subject to } Y + M = \underline{G} \quad (32)$$

where \underline{G} is national income and the notation for the functions $Q(E)$ and $E(A,M)$ is suppressed. The first-order conditions imply

$$U_X X_Y + U_R R_C C_Y = U_X X_M + U_R (R_C C_M + R_M). \quad (33)$$

The private and public budget constraints $X + C = Y$ and $Y = \underline{G} - M$ imply $X_Y + C_Y = 1$, and $X_M + C_M = -1$. Substituting these into Equation 33 results in

$$U_X - (U_X - U_R R_C) C_Y = -U_X - (U_X - U_R R_C) C_M + U_R R_M . \quad (34)$$

Private optimizing decisions ensure that the expressions within the parentheses are equalized to zero, so that public managerial decisions result in the condition

$$R_M = 2(U_X / U_R) . \quad (35)$$

This condition relates public environmental management costs, at the margin, to private willingness-to-pay for recreational experiences.

This framework can be used to explore the effects of changes in ambient concentrations on public welfare, and to estimate the offsetting management outlays required to keep total welfare at its initial level, taking into account private optimizing behavior of the kind analyzed above. Differentiating public welfare with respect to changes in ambient quality and setting the resulting dZ/dA equal to zero

$$U_X X_M + U_R (R_C C_M + R_M) M_A + (U_X X_Y + U_R R_C Y_A) Y_A + U_X X_A + U_R R_A . \quad (36)$$

The two first-order conditions and the relation $M_A + Y_A = 0$ result in the elimination of the first two terms in this expression on the right-hand side, leaving

$$0 = U_X X_A + U_R R_A . \quad (37)$$

Substituting Equation 35 as a final step leads to the desired expression:

$$X_A = -2(R_A/R_M), \quad (38)$$

where, in terms of the notation used earlier

$$X_A = -2(R_E E_A/R_E E_M) = -2(E_A/E_M). \quad (39)$$

This condition states that the compensating willingness-to-pay, in terms of private consumption expenditure, to keep welfare constant in the face of a change in toxic concentrations, can be measured by the change in public management costs needed to offset the physical changes inflicted on natural populations and keep those populations unchanged.

This result provides a firmer justification in economic theory for an idea with a long legal and administrative history. The common law public trust doctrine asserts that State governments have a responsibility to manage and maintain natural resources as guardians of the public interest (Sax). The Federal Fish and Wildlife Coordination Act requires that effects on wildlife and natural populations be considered in the planning of all federally sanctioned projects, and that responsible agencies prepare mitigation plans to offset any losses to habitat or population. Such mitigating programs may include the acquisition and management of additional conservation lands for habitat, restocking, and other measures (Veiluva). Thirty States have specific legislation providing for the recovery of damages from private parties for wildlife losses due to pollution, and, the most common method of valuation is the replacement cost of the

fish and wildlife destroyed (Halter and Thomas). At the federal level, the Clean Water Act provides for the recovery of costs from private parties for restoring or replacing resources lost to pollution, as does the Superfund legislation.

Public management programs for fish and wildlife programs are substantial and diverse, in terms of the numbers of species managed, the variety of management methods, and the levels of expenditure. To take a single example, the list of species under active management by the Division of Fisheries and Wildlife in the Commonwealth of Massachusetts, a State not particularly noted for its wildlife or wildlife recreation, is given in Table 1. The list is substantial. In addition, other species are implicitly managed under ecosystem preservation programs. Management methods include regulation of hunting, fishing and trapping, reservation and improvement of habitat, protection of breeding, nesting and spawning areas, control of predators, operation of hatcheries and restocking of areas, and others. Agencies like the Division of Fisheries and Wildlife in Massachusetts routinely make decisions regarding the allocation of budgetary and manpower resources among programs, the limits on harvesting and the desirability of increasing or reducing population levels. In so doing, they make judgements balancing the demands of various kinds of recreational users, both in the present and, through guardianship of wildlife resources, for the future.

Table 1. Species Managed by the Division of Fisheries and Wildlife, Commonwealth of Massachusetts

<hr/>	
FISHERIES	Mink
	Skunk
<u>Salmonid Program</u>	Otter
	Bobcat
Stocked Trout	
Native Trout	<u>Migratory Gamebird Program</u>
Sea Run Brown Trout	
Lake Trout	Rails
Landlocked Salmon	American Coot
	Woodcock
<u>Warmwater Gamefish Program</u>	Snipe
	Crow
Black Bass	
Esocid	<u>Non-Migratory Gamebird Program</u>
Catfish	
	Ruffed Grouse
<u>Warmwater Panfish Program</u>	Bobwhite Quail
	Pheasant
Panfish	
	<u>Falconry Program</u>
<u>Anadromous Fishes Program</u>	
	Falconry
Atlantic Salmon	
American Shad	<u>Waterfowl Program</u>
	Canada Goose
WILDLIFE	Snow Goose and Bran
	Mallard
<u>Big Game Program</u>	Black Duck
	Wood Duck
Wild Turkey	Canvasback and Redhead
Black Bear	Seaducks
White-Tailed Deer	Mergansers
<u>Small Game Program</u>	NONGAME AND ENDANGERED SPECIES
Cottontail Rabbit	<u>Endangered Species Program</u>
Snowshoe Hare	
Blacktailed Jackrabbit	Shortnose Sturgeon
Gray Squirrel	Plymouth Red Bellied Turtle
	Bald Eagle
<u>Furbearer Program</u>	Peregrine Falcon
	Indiana Bat
Opossum	
Beaver	<u>Nongame Program</u>
Muskrat	
Eastern Coyote	Great Blue Heron
Red Fox	Osprey
Gray Fox	Terns
Raccoon	Purple Martin
Fisher	Common Loon
Weasels	

Moreover, management agencies have substantial experience in formulating mitigation plans under the FWCA and other legal authorizations. Judgements are made on the extent of expenditure warranted to restore habitats which implicitly balance societal benefits against costs. Such agencies have the capability and the experience to estimate the approximate costs of measures to offset the damages caused to natural populations by pollution. These estimates can be based on the costs of management programs which, in many situations, they already have substantial experience with. Therefore, since there is also justification in welfare economics for estimates of willingness-to-pay based on mitigating costs, this provides a potentially simple feasible way to obtain preliminary estimates of potential damages, or, at least, of upper bounds to potential damages.

VI. ESTIMATION OF WILLINGNESS-TO-PAY FOR NONCONSUMPTIVE WILDLIFE BENEFITS USING THE WEAK COMPLEMENTARITY ASSUMPTION

It was explained earlier how the use of the weak complementarity assumption allows consumer surplus from a change in environmental quality to be estimated from observed changes in the demand for a marketed commodity, when that marketed commodity is an input, along with environmental quality, in the production of a wildlife-related recreational experience. (See Figure 6 and accompanying discussion.) The approach can be extended to cover the case when the weak complementarity assumption applies to demands for a group of commodities, all of which must be zero before the consumer is indifferent to changes in environmental quality (Mills and Feenberg).

In the 1980 National Survey of Fishing, Hunting and Wildlife-Associated Recreation, designed by the US Fishing and Wildlife Service, a questionnaire was included for the first time on Nonconsumptive Uses, covering wildlife observation, photography, and feeding. Data were collected on trips made for this kind of recreation, days spent in these activities, and expenditures related to them for equipment, supplies and expense. The nationwide sample consisted of 6,000 individuals. This survey is the basis for estimates of the value of improvements in environmental quality to nonconsumptive recreational users, focusing on activities involving non-game bird populations.

The basic estimating equation is analogous to Equation 31, on the assumption that the quality of the environment on which

the birdwatcher's recreational experience depends is affected by changes in the variety and abundance of bird populations. Greater numbers and diversity of birds enriches the birdlover's experience, and leads to increased demand for the marketable inputs, like time and physical paraphernalia, that go into the "production" of the recreational activity. It has been shown that, depending on whether physical paraphernalia can or cannot be substituted for time in this process, value can be estimated by focusing on a) either time or physical inputs, or b) both time and physical inputs. If variations in the demand for these inputs can be estimated as functions of variations in bird species diversity and abundance, then willingness-to-pay for changes in natural populations can be derived.

The preferred source of data on bird populations is the Department of Interior's annual breeding bird survey, which, on a sampling basis, conducts a count of all birds along specified routes across the nation on a day in June. Indices of diversity and abundance are tabulated and published for ninety-five ecological zones, but the data are also available in greater geographical and species disaggregation. Naturally, geographical variation in bird populations across the United States is considerable. Since the National Survey of FH & WAR is also national in scope, and the geographical locations of respondents are recorded, the two data sources can be combined.

The measures of species abundance and diversity are hypothesized to be determinants of demand for private inputs to bird-watching activities. Household demands are estimated, making use of the geographical variation in bird populations.

The areas between estimated demand curves for specified changes in bird populations can then be used as measures of willingness-to-pay.

A slight modification must be made because the non-consumptive user's questionnaire does not collect price-quantity data for marketable inputs, but rather expenditure data, for the most part. This means that what can be measured is the increase in expenditures as a function of the improvement in the quality of the natural environment. In Figure 6, this is not the shaded area between the demand curve, but the increase in area beneath the demand curve as input use rises from C_0 to C_1 . However, this is not an insuperable problem. It is readily established that for constant elasticity demand functions, consumer surplus is related to expenditures by a simple formula:

$$M = \frac{1}{e-1} E_x, \quad \text{for } e > 1 \quad (40)$$

where M is Marshallian consumer surplus, e is the price elasticity of demand, and E_x is the amount of expenditure. For price elasticities less than one, Marshallian consumer surplus is infinite in constant elasticity systems. Therefore, if the change in expenditures is known as a function of changes in environmental quality, the change in consumer surplus is

$$M(Q_1) - M(Q_0) = \frac{1}{e-1} (E_x(Q_1) - E_x(Q_0)). \quad (41)$$

The assumption of a constant price elasticity is not serious, but must be interpreted as an average or arc elasticity over the

entire range of demand. The specification of non-constant elasticity is not preferable, since estimated point elasticities cannot be extrapolated outside the range of the data, which is obviously required in the system illustrated by Figure 6. An estimate of the arc elasticity must be derived exogenously.

A related approach is to specify the production function for non-game birds in terms of concentrations of toxic substances, availability of habitat, and other relevant variables, and then to use geographically varying estimates of those determinants in estimating demands for recreational inputs. This is equivalent to substituting the biological production function $E = E(A, M)$ into the demand equation 31. Data exist by State, and even more disaggregated regions, on availability of habitat, on pesticide usage per unit area, on toxic concentrations in monitoring media, on industrial activities and emission loadings, and other variables that would enter into the biological production function. This approach would provide estimates of the direct effects of toxic concentrations and other environmental variables on recreational demands, and could lead to a direct estimate of willingness-to-pay for reduced concentrations by this category of consumers. The alternative approach, yielding estimates of willingness-to-pay for enhanced bird populations, produces an intermediate product in the estimation of benefits from toxic substances control, but one which can potentially be used in a variety of contexts. In reality, the two approaches are complementary.

VII. NONMARKET APPROACHES TO THE ESTIMATION OF TOXIC DAMAGES

There is another approach to the problem of estimating damages from toxic effects, based on the use of nonmarket data derived from carefully constructed direct surveys of consumer willingness-to-pay for precisely defined environmental benefits. This approach has been developed substantially in recent years, based largely on the work of David, Randall, Brookshire, and Schulze (Schulze, d'Arge-Brookshire).

Attempts to ascertain willingness to pay for increments or to forestall decrements in environmental quality, or for other public goods, by direct questioning had long been discouraged by theoretical arguments that consumers would tend to misstate their true preferences in one direction or the other, depending on their perceived strategic interest, because of their knowledge that they would not be excluded from the enjoyment of the public good, whatever their stated willingness to pay. Considerable evidence has now been accumulated that this "strategic bias" in responses is of little actual importance, although other sorts of bias may indeed be significant.

The direct survey, or "contingent valuation" approach has been shown to produce plausible estimates of willingness to pay for a variety of environmental services: health and visibility

benefits from reduced air pollution; recreational benefits from improved water quality; increased stream flow and others. The advantages peculiar to this approach are:

- it is feasible when it is impossible or impractical to construct estimates from market data, as in the case of unique prospective changes in environmental quality, concerning which there are not yet any relevant market transactions:
- it is relatively inexpensive, on the scale of household survey costs;
- it can capture categories of benefits that do not appear in market data, such as the willingness to pay of individuals who are not users of the environmental amenity in question, either directly or indirectly.

The latter point deserves elaboration. That there are people willing to pay for environmental quality who are not, and will not be, direct consumers of the amenity in question, has been well established. People contribute to save the cheetah and the rhinoceros who will never see one in the wild. Probably, there are many who are sufficiently upset at the knowledge that pesticide residues appear in arctic birds to be willing to pay something to improve the situation, even though they will never visit the arctic. Such individuals display preferences and demand functions that violate the weak complementary assumption: the marginal utility of an environmental improvement is positive although any set of consumption levels (such as those reflective of direct use of the amenity) may be zero. The "existence value" of environmental amenities to such vicarious consumers has been

shown to be substantial and widespread. (Brookshire et al.)

Similarly, option value, the difference between the total willingness to pay for the assured enjoyment of an amenity and the expected compensated consumer surplus, given uncertainty as to future demand, cannot be investigated but by the use of nonmarket data. (Bishop)

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PART 5

THE ASSESSMENT OF ECOLOGICAL HAZARDS FROM PESTICIDES: THE USE OF QUALITATIVE MODELLING IN DECISION ANALYSIS

Robert Repetto
Anthony C. Janetos

I. INTRODUCTION

The Federal Insecticides, Fungicides and Rodenticides Acts (FIFRA) directs the EPA administrator to deny or to restrict the registration of a chemical if he judges that, in normal use, it could cause "unreasonable adverse effects on the environment," defined in the act as "any unreasonable risk to man or the environment, taking into account the economic, social and environmental costs and benefits of the use of any **pesticide.**"¹ The legislation underlying pesticides regulation mandates a balancing of benefits and costs. The current system of testing and pre-regulatory analysis in the pesticides program cannot do this. Producers of pesticides must apply to the EPA to register them for their intended uses (or re-register them, if already registered under looser guidelines in force before passage of the 1972 Federal Environmental Pesticide Control Act amending FIFRA), submitting with the application a variety of data on the properties of the chemical and toxicity test results. Much of the data is intended to enable EPA to assess the behavior, movement, and fate of the chemical through the environment. The testing **regimen**² for ecological hazard consists of short and long-term toxicity tests on birds, wild mammals and aquatic

organisms. These tests are organized sequentially and proceed from laboratory tests to applied field tests. Only acute toxicity tests are routinely required for all chemicals. Longer-term toxicity tests, such as those on interference with reproduction, are required if the chemical is persistent, bioaccumulates, or would be used in such ways as to expose populations repeatedly or continuously. Simulated or actual field studies are required only **infrequently**.³ Tests for adverse effects on nontarget insect and plant species are not routinely required. In general, the application of this tiered testing system is such that if likely exposures are well below acute toxicity levels and the chemical's properties and uses are not such as to give rise to suspicions of long-term adverse effects, ecological testing is not pursued beyond the basic short-term laboratory tests.

The likelihood of errors in the predicted ecological effects of chemicals based on this testing regimen is probably high, and so, consequently, is the likelihood of errors in regulatory decisions. Both false positives and false negatives occur. A recent National Research **Council**⁴ review of the subject indicated some of the reasons. The NRC review expresses special concern at the failure of the current testing regimen to consider ecological interactions among species, in assessing the risks of chemicals to populations. It concludes:

"Single species tests can provide much information on the concentrations and durations of exposures to chemicals that result in changes in survival, reproduction, physiology, biochemistry and behavior of individuals within particular species, but results from such tests cannot predict ... impacts beyond this level of biological **organisation**."

Within a single, density-dependent species, an increase in mortality may or may not affect population size. Within an ecological system of interacting species, a toxic effect on one species might not affect its own equilibrium population level, but might affect that of its predator or prey. Such possibilities are not encompassed within current testing or analytical frameworks used for hazard assessment.

The EPA, while aware of the limitations of current methods of ecological hazard assessment, is reluctant to require additional testing and data requirements. Testing, especially outside the laboratory, is complicated and costly. Moreover, there is uncertainty as to the kinds of additional information required to reduce uncertainties about environmental hazards, and the capacity of any particular set of tests to do so.

Two general issues emerge from this brief discussion: first, how to regulate pesticides and other chemicals more efficiently in the face of uncertain ecological risk; and second, how to make better decisions about testing for ecological effects. The methodological approach presented in this paper indicates how incomplete information on ecological effects can be used to make judgements on the cost-effectiveness of further tests and on regulatory options. Specifically, it demonstrates a simple analytical technique with which qualitative information on ecological structures can be used within a decision analysis framework to improve predictions of risk based on direct toxicity

tests performed on individual species. The paper is expositional. It uses a simple problem, structurally akin to but less complex than actual regulatory problems, to explain the approach and demonstrate its potential usefulness.

II. TECHNIQUES

The Elements of Decision Theory⁶

The premise of decision analysis is that the decision maker wishes to maximize expected net benefits when choosing among options in the face of uncertainty regarding their consequences. Benefits may be defined in terms of money or other measures of payoff. The value of additional information about the consequences of actions is the difference between the expected net benefits of decisions made with the additional information and the expected net benefits of decisions made without it. This value can be compared with the cost of obtaining the additional information.

A decision problem has four general components: (1) the set of i decision options, a_i , which can include in the regulatory context, both regulatory options and testing options; (2) the set of j possible states of nature, s_j , that determine the possible consequences of the actions; (3) the joint probability distribution of the possible states of the world $f(s_j)$, which is regarded as a subjective probability distribution; (4) the set of payoffs for each action, $U(a_i | s_j)$, given a particular state of the world. For example, a possible state of the world might be that a certain pesticide is toxic to fish, a possible action might be to register it for a set of agricultural uses, and the payoff might be the ecological damages that would result from registration, given the chemical's toxicity.

The decision criterion is then assumed to be to select that option which maximizes the expected payoff, $E(U)$, expressed as:

$$\max_{a_j} E(U) = \sum_j U(a_j | s_j) f(s_j) \quad (1)$$

Since this is the decision process which presents itself prior to any further testing or information gathering regarding consequences, the distribution $f(s_j)$ is conventionally called the prior probability distribution.

If additional information is obtained, these probabilities might have to be revised. A new set of probabilities, $f(s_j | m)$ is adopted, after receipt of the information m , signifying the likelihood of s_j being the state of the world, given that information m has been obtained. A method of deriving this posterior probability is the use of Bayes theorem, which states:

$$f(s_j | m) = f(s_j) f(m | s_j) / \sum_j f(s_j) f(m | s_j) \quad (2)$$

The posterior distribution can be seen as a weighted average of the prior probabilities, the weights being the likelihood of receiving information m were s_j , the actual state of the world. If some messages were very unlikely to even impossible in some possible state of the world, it is clear that the posterior probabilities might differ substantially from those held prior to the new information.

The expected value of additional information is then the difference between expected payoffs when options are chosen using revised probabilities and original probabilities:

$$\text{expected value of information } m = \max_{a_i} E(U_m) - \max_{a_i} E(U) \quad (3)$$

where $E(U_m)$ represents expected payoffs evaluated under revised probabilities, given information m . Conceptually, it is this value which should be compared with the cost of obtaining the information. It is clear that rational decisions regarding testing require not only information on the costs of the tests, and their power, but also estimates of the magnitude of gains and losses of alternative actions in possible states of the world.

Qualitative Analysis of Complex System

Ecosystems may be characterized as collections of species, as fluxes of materials and energy, or as structural objects, e.g., 'forests' or 'deserts'. Each characterization may be useful in a particular context. However, all are descriptive and have limited predictive power. A more analytical representation of an ecosystem considers it as a collection of interacting populations of animals and plants. The simplest representation describes the growth of each population as a differential equation:

$$dN/dt = f(N) \quad (4)$$

where N is the number or mass of a species and $f(N)$ is a general function that describes the relationship between the population's size and its growth rate.⁷ However, since no population actually

exists independently of others, each population's growth must be represented as:

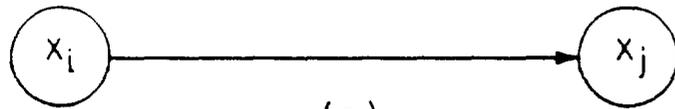
$$dN_i/dt = f_i(N_i, C) \text{ for } i = 1, 2, 3, \dots, n \quad (5)$$

where n is the number of species in the system, C is a representation of non-biotic inputs, and other symbols are as before.

In hazard assessment there is neither time nor resources to make a full study of the dynamics of threatened ecosystems. Analytical techniques are required that do not depend on the precise quantitative specification of the population growth equations.

Loop analysis, introduced to biology by Richard Levins⁸, and extended by Lane and Levins⁹, **Lane**¹⁰, Puccia and **Levins**¹¹, and **Puccia**¹² is such a technique. It is a graphical method of utilizing qualitative information about the interactions in a system. Labeled vertices in the graph represent the populations in the system. The interactions between populations are represented by line segments between vertices, each of which describes the marginal effect of a change in one population's size on the growth rate of the other.

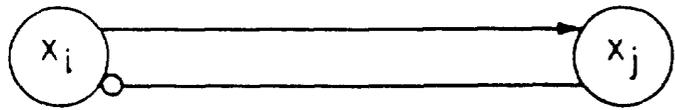
Figure 1 illustrates all the basic relationships between two populations. A line segment originating from population i and ending in an arrow on population j means that an increase in i results in an increase of j (Figure 1a). This can be written as an interaction coefficient: $a_{ji} = +1$.



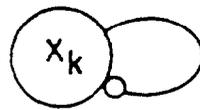
(a)



(b)



(c)



(d)

Figure 1. All the Basic Relationships Between Two Populations

A segment arising from j and terminating on i with a small circle indicates that a rise in j 's population decreases the population of i (Figure 1b): $a_{ij} = -1$.

Figure 1c shows the most common relationship between two populations in these diagrams: species j preys on species i . The dynamics are captured in the diagram: an increase in j results in a decrease in i , and an increase in i results in an increase in j . This situation also illustrates well the concept of local stability. Small increases in the population of species i lead to an increase its predator j , with the result that i is forced back to its original equilibrium. Thus, fluctuations in the system are eventually damped out and the system is locally **stable**.¹³

The last basic relationship is illustrated in Figure 1d. There is an equilibrium population size of species k set by environmental parameters, and any deviations are self-limiting. In the notation of interaction coefficients, $a_{kk} = -1$.

Loop analysis provides a way of using these qualitative relationships among populations to make predictions about the direction of change in populations of interest of perturbations elsewhere in the system. It properly describes only those systems that are at or near dynamic equilibrium (i.e., those in which populations' average sizes are roughly constant over time), and perturbations from equilibrium which are small. Thus, its usefulness for ecological risk assessment lies in exploring sub-acute, chronic effects of pollutants on ecosystems, and in predicting the indirect effects of pollutants on species to which they are not directly toxic.

Loop analysis provides answers to two questions: (1) Is the system dynamically stable, and (2) Will populations of interest grow or diminish, due to changes elsewhere in the system? What follows is a description of how to do loop analysis, using the system of Figure 2. The exact computational recipes, abstracted from Levins¹⁴ and Lane and Levins¹⁵ are given in the appendix.

Figure 2 represents a three-species system: a fish, F, that eats zooplankton, Z, that eat phytoplankton, P. The phytoplankton exhibit self-limiting growth. To ascertain first whether this system is dynamically stable requires an understanding of the concept of feedback.

Feedback is the effect of one component of the system on its own growth. Component is loosely defined; it may be a single species or it may be combinations of species that are linked together in loops. A loop is defined as a closed series of linked populations in which each vertex is visited once. A component may also be a combination of loops and single species or loops and other loops. In order for a system to be stable, the feedback at each level must be negative: i.e., each component must show self-limiting growth, whether the component is a single species, a loop of 2 species or a loop of n species. In the simplest case, the feedback of a loop of n species is the product of the interaction coefficients of the links in the loop. The sum of the feedbacks of all loops of length n is the total feedback of the system at level n. More complicated rules for combinations of loops are in the appendix.

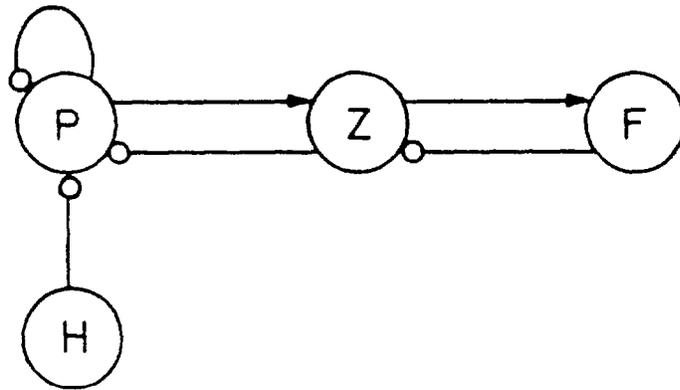


Figure 2. Representation of a Three-Species System:
a Fish (F), That Eats Zooplankton (Z),
That Eats Phytoplankton (P)

In Figure 2, the feedback at level 1 is negative because P shows self-limiting growth, and Z and F make no contribution. The feedback at level 2 is negative, because both possible two loops have negative values. The feedback at level 3 is a combination of a two-loop (Z and F) and a disjunct one-loop (P on itself), and is also negative (see Appendix). Thus, this system is locally stable.

The second question is whether populations will grow or shrink in response to a perturbation, such as the input of a chemical directly toxic only to the phytoplankton, P. Loop analysis explicitly takes into account the fact that the effect of a perturbation on a population is the result not only of the direct effects from the point of entry into the system, but also of the indirect effects that stem from the rest of the system.

Loop analysis does this by considering both the direct pathways of effects and the complements of those pathways. A path between A and B is the collection of links that begin at A and end at B, visiting each intermediate population only once. Like a loop, a path is directional, and its value is the product of the interaction coefficients of its component links. Unlike a loop, a path is not a closed system. The complement of a path is the set of populations that are not on the path.

The direction of the effect on a species B as a result of input to species A depends on the product of two things: the value of the direct path from A to B and the highest level feedback of the complement of the path. The complement of each path can be thought of as a black box containing the rest of the ecosystem; its feedback is the effect that the rest of the

ecosystem has on the path. This must be divided by the feedback of the entire system. Since in large systems there may be more than one direct path from one species to another, we sum the products of each path and its complement.

Applying this to our example in Figure 2 yields the following:

1. The direct effect of the herbicide on the phytoplankton is negative, by definition. The complement of the input is the two-loop of zooplankton and fish, which has negative feedback, as does the whole system. Thus, the phytoplankton population decreases, i.e., changes in the same direction as the input.
2. The zooplankton's population level remains unchanged. The direct path from phytoplankton to zooplankton is positive, indicating that the two populations should change in concert, but the complement of the path is population F, which does not have self-limiting growth, hence has no feedback, and negates any change in Z.
3. The fish's population falls. The direct path from phytoplankton to fish has a positive sign, and the complement is defined to have negative feedback; thus the fish and the phytoplankton change in concert. The definition of complement in this case is an algebraic convenience: when a path from one species to another includes all the species in the system, the complement of that path is simply defined to have feedback of -1 (see Appendix).

These results are intuitively reasonable as well. The net population size for each population depends on the direct effects from the herbicide input and the balancing effect from the rest of the system. For the zooplankton, the net result is that any change from below is counteracted by its predator, and the zooplankton's population size remains unchanged. The complements of the paths leading to the phytoplankton and fish reinforce the direct paths, and both populations decline.

An important point to note is that the predictions of loop analysis may differ from those generated by single-species toxicity tests. Herbicides are not toxic to animals, yet this simple system leads to the prediction that fish populations will fall. It is precisely this type of prediction, springing from knowledge of the underlying interactions, that makes loop analysis valuable.

In more complicated systems, this method does not always lead to unambiguous qualitative predictions about the direction of effects. Even in such cases, it may provide guidance as to the relative likelihoods of possible outcomes. It may also indicate the linkages in the system on which resolutional testing which would resolve the ambiguity. By the same token, it may indicate what unknown quantitative parameters of the system are essentially irrelevant to the evaluation of the risks under consideration.

III. APPLICATION TO A SIMULATED PESTICIDE PROBLEM

This section uses a hypothetical problem involving testing and regulation of a pesticide for ecological hazard to show how qualitative analysis and decision theory complement each other. The point of departure is the information available from direct toxicity testing. It is assumed that a chemical is tested which proves toxic in chronic exposure to only one fish species, a zooplankton feeder F^1 , but not to herbivore nor carnivore species F^2 and F^C , nor does it affect phytoplankton nor zooplankton growth. The direct regulatory problem is whether or not to restrict the chemical's regulation on the basis of these test results. It is assumed that the approximate annual value of economic losses that would result from injuries to each of the fish stocks has been estimated. While this is not commonly done in the regulatory analyses underlying pesticide registration procedures, there are methods for establishing approximate **magnitudes**.¹⁶ Similarly, it is assumed that the annual costs, in terms of foregone benefits to pesticide users if the chemical's registration were restricted, have been estimated. Hypothetical values have been assigned to these costs and benefits, as indicated in Table 1a.

Given these values, the regulatory decision can be resolved once judgements regarding the degree of risk have been made. For reasons noted in Section I, the lab results do not provide highly reliable predictions of field experience. there is considerable probability of false positives and negatives. It is assumed that

Table 1. Cost and Damage Values / Damage Probabilities

A. Cost and Damage Values Assumed for Decision Analysis:

Damage to F^1	40
Damage to F^2	180
Damage to F^C	240
Cost of regulation	100
Costs of qualitative information on ecosystem structure	10
Costs of quantitative information on ecosystem functioning	10

B. Damage Probabilities: Prior and Posterior to Qualitative Analysis of Each Ecosystem:

	Prior	System <u>1</u>	System <u>2</u>	System <u>3</u>
$P(O_1)$: Damage to F^1 only	.0469	.00	1.00	.45
$P(O_2)$: Damage to F^2 only	.4219	.50	.00	.10
$P(O_3)$: Damage to F^1 and F^2	.1406	.00	.00	.15
$P(O_4)$: Damage to none	.0156	.00	.00	.30
$P(O_5)$: Damage to F^1 and F^C	.1406	.00	.00	.00
$P(O_6)$: Damage to F^2 and F^C	.0469	.00	.00	.00
$P(O_7)$: Damage to all	.0469	.00	.00	.00
$P(O_8)$: Damage to F^C only	.0469	.00	.00	.00

these probabilities of error are set at one in four, .25, for fish species, and that, in the absence of information regarding ecosystem structures, these probabilities are taken as independent. This leads to the set of joint probabilities for the eight possible outcomes listed in the first column of Table 1B. Since the damage estimates are taken to be independent and additive, these probabilities can be used to calculate expected losses if registration is not restricted. These expected losses can be compared to the estimated costs of regulation, and the preferred action, on the assumption of no further testing, can be selected. This corresponds to the procedure indicated in equation (1) above.

However, additional information can be obtained at a cost. This consists of qualitative information about the structure of ecosystems into which the pesticide would be introduced, and quantitative information about the interaction of species within these systems. Qualitative information consists only of knowledge of (a) the species present in each system affected by the pesticide, (b) the relative frequencies of these systems, and (c) the signs (positive, negative, or zero) of the linkages between species in all systems. Quantitative information consists of data on the magnitudes of the interactions (a_{ij}) between species. Therefore, two further regulatory decisions can be made, prior to the decision to restrict regulation or not: first, the decision whether or not to bear the cost of finding out the qualitative structure of relevant ecosystems and their

frequencies: second, the decision whether then to go further and investigate the quantitative structure of those systems prior to the regulatory decision.¹⁸

In summary, the decision problem is to choose among the following possibilities, based on the results of the lab test data:

- a) register the pesticide for its proposed uses;
- b) restrict the pesticide's registration;
- c) investigate the relevant ecosystems qualitatively, and then decide whether or not to regulate;
- d) investigate the systems qualitatively and then quantitatively, and then decide whether or not to regulate.

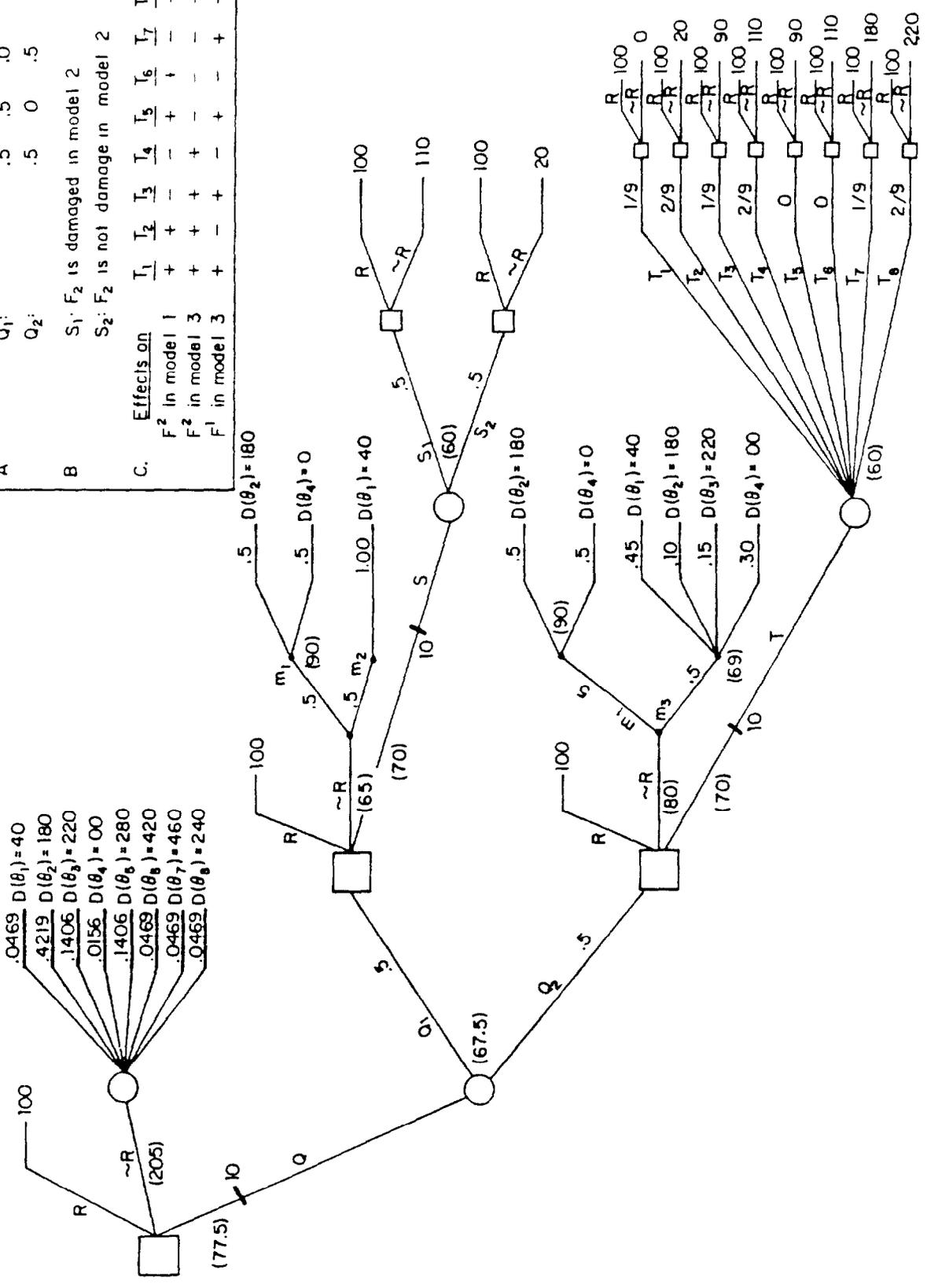
The decision process is outlined in a "tree" format in Annex I. It is clear from the top part of the tree diagram that, in the absence of further testing, the regulatory decision based on lab data would be to restrict pesticide regulation: largely because of the probable damages to type 2 (zooplankton-feeding) fish, the expected losses from non-regulation would be 205, compared to regulatory costs of 100. This corresponds approximately to the current decision process. The options for further testing change the decision outlook considerably.

It is assumed that the relevant ecosystems are of three possible types, diagrammed in Figure 3(a)-Figure 3(c). There are two mutually shading phytoplankton species, P^S and P^R , which are grazed by zooplankton, Z , and an herbivorous fish, F^1 . The

CODE			
Frequencies of			
	m_1	m_2	m_3
A	.5	.5	.0
B	.5	0	.5

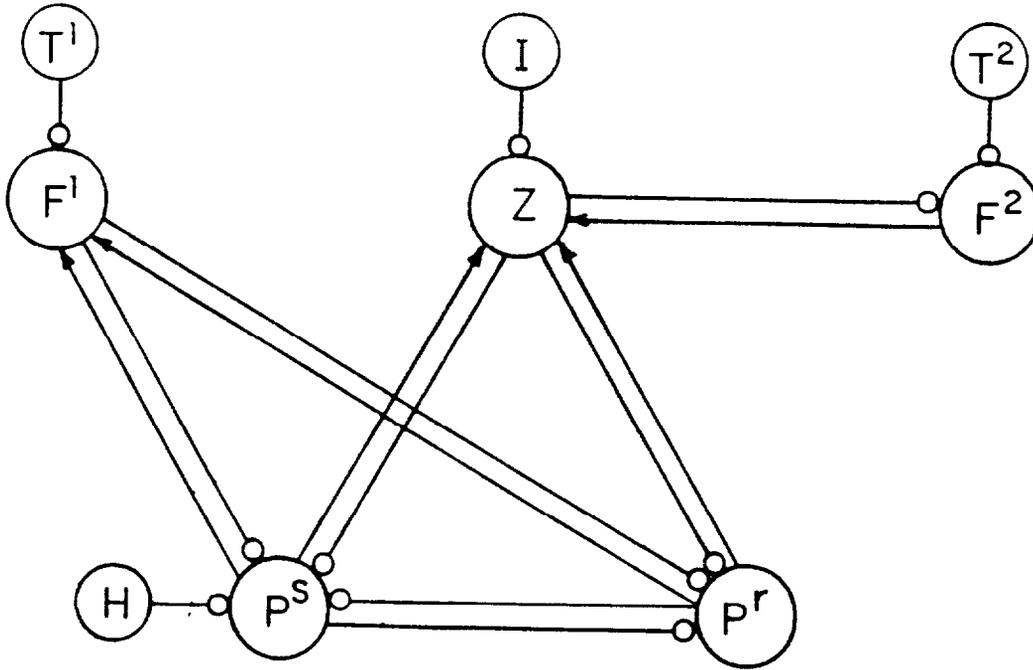
Effects on								
	I_1	I_2	I_3	I_4	I_5	I_6	I_7	I_8
F ² in model 1	+	+	-	-	+	+	-	-
F ² in model 3	+	+	+	+	+	-	-	-
F ¹ in model 3	+	-	+	-	-	+	-	+

S₁: F₂ is damaged in model 2
 S₂: F₂ is not damaged in model 2



Annex 1. Decision Tree for Regulatory Analysis

a. System one



b. System two

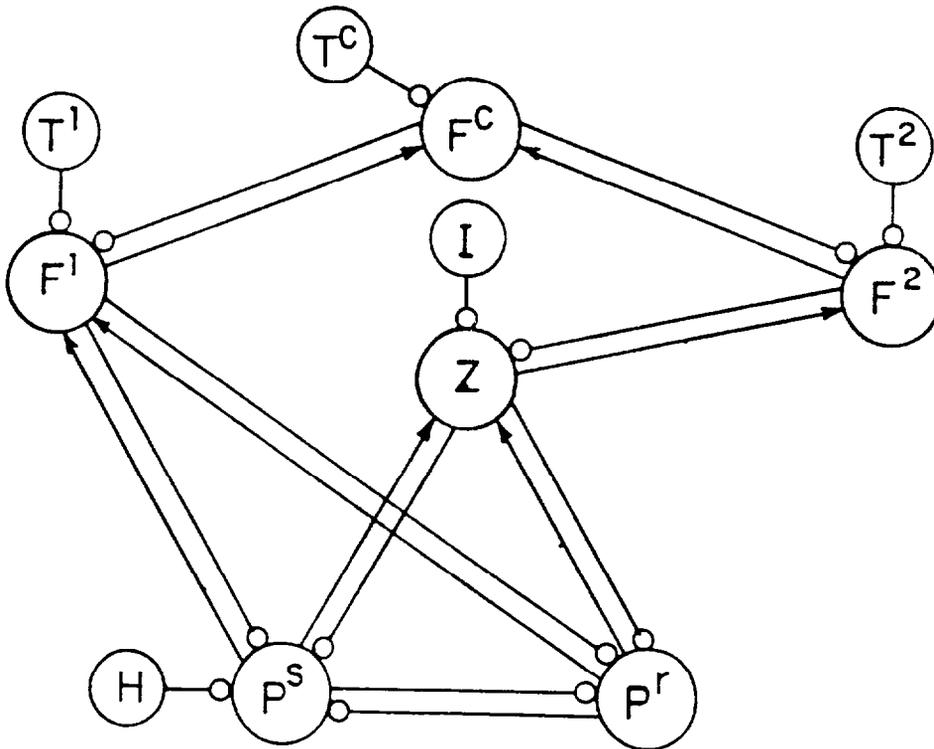


Figure 3. Loop Diagrams of Ecosystems

c. System three

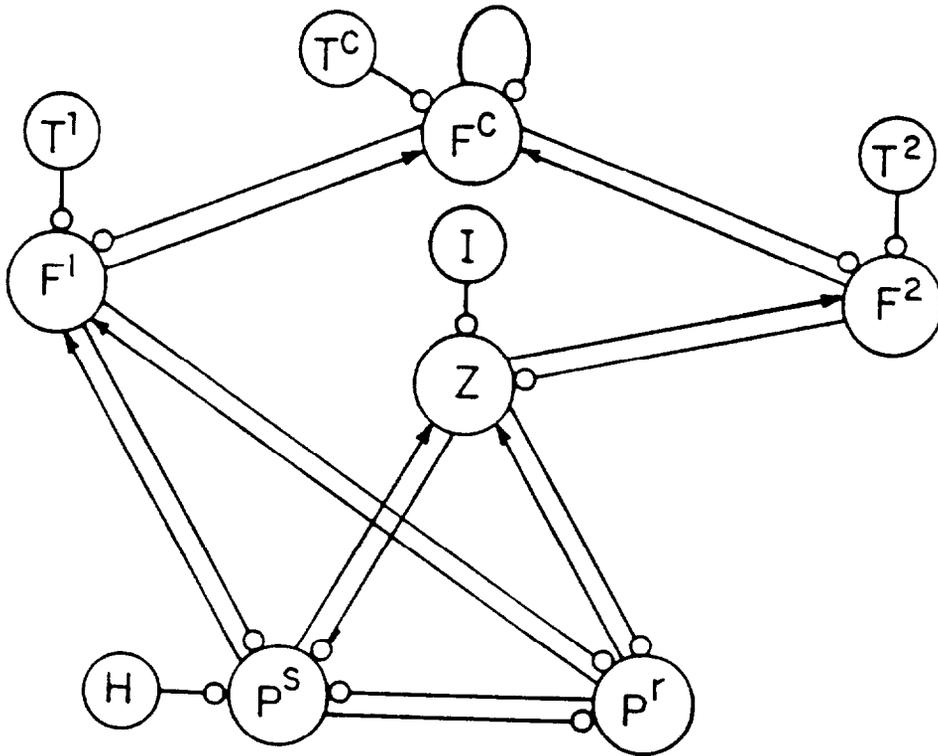


Figure 3 (continued)

zooplankton are preyed on by a fish, F^2 . In systems 2 and 3 there is also a carnivorous fish, F^C , which preys on both F^1 and F^2 . This species may or may not be self-damped, as represented in systems 3c and 3b respectively.

Table 2a-2c provides provides a complete summary of the results of qualitative analysis of effects on the fish in these ecosystems, not only for the pesticide under analysis, T^2 , which is toxic to F^2 , but also for other possible chemicals' impacts: that of a pesticide, T^1 , toxic to the herbivorous fish; that of a pesticide, T^C , toxic to the carnivorous fish; that of an insecticide, I, toxic to the zooplankton; and that of an herbicide, H, toxic to one susceptible phytoplankton species. A plus sign (+) indicates that the overall predicted impact would be to increase the equilibrium stock of the population; a minus sign (-) indicates a predicted decrease; and a question mark indicates ambiguity. A question mark followed by a plus or minus sign indicates that qualitative analysis leads to a greater likelihood, but not a definite prediction, of the indicated effect, a result which will be explained below.

This table shows how qualitative analysis can alter substantially the prior probabilities of hazard based on lab testing. Looking first at the predicted effects, $\partial F^1(t)/\partial T^1$, $\partial F^2(t)/\partial T^2$, and $\partial F^C(t)/\partial T^C$, which are the overall effects of chemicals on the species to which they are directly toxic, we see the possibility of false positives. Only in the case of F^C , representing the highest trophic level in the systems, is the adverse effect predicted by lab tests expected to be the same when ecological interactions are taken into account. Chemicals

Table 2. Effects of Pollutants on Species Populations

	2a)		2b)			2c)		
	F ¹	F ²	F ¹	F ²	F ^C	F ¹	F ²	F ^C
T ¹	+ ^a	- ^a	+	-	?(+) ^b	+	-	?(+)
T ²	+	? ^a	-	+	+	?(-)	?(-)	+
T ^C	n.a.	n.a.	?	?(+)	-	?	?(-)	-
I	0	+	-	+	+	-	+	+
H	+	?(-)	?	?(+)	?	?	?(+)	?

a) A plus sign indicates that $\partial F^i(t)/\partial T^j > 0$; a minus sign that $\partial F^i(t)/\partial T^j < 0$; a question mark that the sign of $\partial F^i(t)/\partial T^j$ is ambiguous.

b) A question mark followed by a plus (or minus) sign indicates that a positive effect (negative effect) is likelier but not necessary.

directly toxic to species F^1 and F^2 respectively would not necessarily be expected to result in reductions in population levels; the predicted effect on F^1 would be to raise equilibrium stocks, because all the feedback runs counter to the direct impacts. $\partial F^2(t)/\partial T^2$, is ambiguous in systems one and three.

On the other hand, the results in Table 2 also illustrate the likelihood that inferences from direct lab toxicity tests would result in false negatives. In all systems, increases in exposures of the herbivorous fish F^1 to the chemical T^1 directly toxic to it would reduce stocks of F^2 , even though there is no direct toxic impact. In system two and possibly in system three, stocks of F^1 would be reduced by the effects of a chemical harmless to it but toxic to F^2 . A chemical, I, toxic to the zooplankton would definitely be expected to have adverse effects on the F^1 population whenever the carnivorous fish is present, although tests would show no direct toxicity. The impacts of an herbicide, H, on fish populations are quite ambiguous without further information, and might be adverse, although by definition the lab test would indicate harmlessness. Therefore, qualitative analysis indicates the possibility of false negative predictions from direct toxicity tests, as well as false positives.

The following paragraphs illustrate the incorporation of these results into decision analysis by pursuing the analysis of regulatory and testing decisions regarding T^2 , the chemical toxic to the zooplankton feeder. The lower branch of the "tree", marked Q for qualitative testing, indicates the possibilities. By assumption, possible occurrences of the three systems are (Q1)

system one half the time and system two half the time; or (Q2) system one half the time , and system three, with F^C self-damped, the other half. These possibilities are much simpler, of course, than those likely to be encountered in actuality, but the underlying idea that a tier of data gathering beyond direct toxicity testing consists of obtaining information about the qualitative structures of ecosystems affected by the chemical of concern and their relative frequencies is adequately represented by these alternatives. They also demonstrate the importance for system behavior of self-damping of species.

It is assumed that this qualitative uncertainty can be resolved at a cost of 10 units. If this information is obtained, the subsequent choices are to restrict registration on the basis of the new data, to decide not to regulate, or to decide to undertake further quantitative testing to resolve ambiguities. Before it can be decided whether this expenditure is worthwhile, the impact of the information on these subsequent decisions must be investigated. Columns two through four in Table 1B show the revision of prior probabilities that results from qualitative analysis of each ecosystem.

If Q1 is the case, the expected cost of not restricting registration appears to be quite different than it does without the information. Table 2b indicates that in system two, only F^1 is harmed; Table 2a shows that in system one, F^1 is unharmed but F^2 may be harmed. Therefore, at the end of the branch of Q1 corresponding to no regulation ($\sim R$), $D(\theta_1)$, occurs half the time with probability one, while the other half of the time, either $D(\theta_2)$ or $D(\theta_4)$ occurs. If, with no further insights with which

to resolve the ambiguity, these possibilities are considered equally likely, leading to the subjective probabilities along the branches, the expected losses from the decision not to regulate are 65 (an average of 40 and 90). These contrast with the expected loss of 205 under prior probabilities. The difference arises because, on the basis of qualitative information, losses to F^C are ruled out and losses to F^2 are deemed much less likely. Thus, if it were known that Q1 were the case, the decision would likely be not to restrict regulation; whereas, on the lab tests alone, it would have been to restrict.

If Q2 were known to prevail, the possible losses from the decision not to regulate would be more complex: in system one, there might be losses to F^2 , as before; in system three, as shown in Table 2c, there might be losses to F^1 and F^2 . However, qualitative analysis provides further insight into these possibilities. In Table 3, Panel A presents the details of the analysis of impacts on species F^1 in system three. There are three pathways by which toxic effects from T^2 reach this species. The first is positive, since the adverse direct effects on F^2 depress stocks of the predator F^C and reduce predation on F^1 . The other two work through increased stocks of zooplankton, subject to less predation from F^2 , and reduced mutual shading between the phytoplankton species P^C and P^r , to raise stocks of F^1 . However, assuming the entire system is stable, so that feedback at level six is negative, panel A shows that the longer paths have the same impact as their signs, while the shorter path has the opposite impact. Consequently, if the quantitative

Table 3. Path-Loop Flow Chart

A. $F^1(t) / T^2$

Path	Sign of Path	Complement of Path	Disjunct Loops	Number of Loops (m)	$(-1)^{m+1}$	Product of Loops: V(L)	$(-1)^{m+1}V(L)$
$T^2 F^2 F^C F^1$	+	$Z P^S P^r$	$Z P^S P^r Z$	1	+1	+	+
			$Z P^r P^S Z$	1	+1	+	+
$T^2 F^2 Z P^r P^S F^1$	+	F^C	$F^C F^C$	1	+1	-	-
$T^2 F^2 Z P^S P^r F^1$	+	F^C	$F^C F^C$	1	+1	-	-

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B. $F^2(t) / T^2$

Path	Sign of Path	Complement of Path	Disjunct Loops	Number of Loops (m)	$(-1)^{m+1}$	Product of Loops: V(L)	$(-1)^{m+1}V(L)$
$T^2 F^2$	-	$F^C F^1 P^S P^r Z$	$F^C F^C; F^1 P^S Z P^r F^1$	2	-1	-	+
			$F^C F^C; F^1 P^r Z P^S F^1$	2	-1	-	+
			$F^C F^1 F^C; P^S P^r Z P^S$	2	-1	-	+
			$F^C F^1 F^C; P^S Z P^r P^S$	2	-1	-	+
			$F^C F^C; F^1 P^S F^1; Z P^r Z$	3	+1	-	-
			$F^C F^C; F^1 P^r F^1; Z P^S Z$	3	+1	-	-

strength of the shorter path dominates that of the longer ones, one would expect the overall impact on F^1 to be negative. Prior to quantitative measurement of these paths and corresponding feedback, it would be plausible to assign a probability of 0.6 to this negative outcome.

Panel B presents the qualitative analysis of the effect of chemical T^2 on the species to which it is directly toxic, F^2 , once interactions are considered. There is only the one direct path, to which all other species in the system are complementary. However, as Panel B shows, the feedback of this complement consists of a number of sets of disjunct loops that involve all the species. Of these, four have positive signs, two negative. It is likely that the positive signs dominate, in which case the overall effect would be opposite in sign to that of the direct path. Thus, despite direct toxicity, it is likely that a positive effect on species stocks would result. A probability of .75 can be assigned to this outcome, and one of .25 to the alternative, that stocks of F^2 would decline.

Taking these probabilities as independent, the joint distributions of the probabilities of various damage levels can be derived: $P(D(\theta_1))$, that only F^1 would be harmed is $.6 \times .75 = .45$; $P(D(\theta_2))$, that only F^2 would be harmed is $.4 \times .25 = .1$; $P(D(\theta_3))$, that neither F^1 nor F^2 would be harmed, is $.6 \times .25 = .15$; and $P(D(\theta_4))$, that neither F^1 nor F^2 would be harmed, is $.4 \times .75 = .30$. The probabilities of all other outcomes are zero, since analysis of system three unambiguously predicts no damage to F^C .

Therefore, expected losses if exposures in system three are not regulated are 69, as shown in the relevant branch of the decision tree, while expected losses in system one are 90. Overall, the expected loss is 80 across both systems. so, whether Q1 or Q2 were actually the case, the regulatory decision based only on qualitative analyses would be not to regulate, whereas without qualitative analysis it would have been to regulate. Before data were obtained on the truth of Q1 or Q2, the expected losses from nonregulation would be $(1/2)(80+65) = 72.5$. The expected savings from the availability of qualitative information about system structure is thus $100 - 72.5 = 27.5$, much above the cost of the information. The preliminary benefit: cost ratio of qualitative analysis is 2.75:1.

However, further decisions remain to be taken. The decision to regulate or not can be deferred, and further quantitative data can be obtained about relevant parameters of the ecosystems. Qualitative analysis indicates what data are most relevant, thus eliminating from consideration a variety of quantitative tests. It is assumed that relevant ambiguity can be eliminated from either branch, Q1 or Q2, at a further testing cost of 10 units. Is it worthwhile to carry testing to this further level, once the qualitative structure of ecosystems are known?

Along branch Q1, corresponding to equally frequent occurrences of systems one and two, the only ambiguity after the qualitative analysis is whether F^2 in system one would be harmed, and this is regarded an even bet prior to quantitative testing. If it would be harmed, expected losses would be

$(1/2)(180+40) = 110$, and the decision would be taken to regulate, at the lower cost of 100. If it were not, losses would be $(1/2)(40) = 20$, and non-regulation would be preferred. Giving these alternatives equal ex ante probabilities, expected losses were the decision taken (not to regulate) on the basis of qualitative information alone, and it would not be cost-effective to pursue testing further. Given the qualitative results, complete elimination of ambiguity is not worth the cost it would entail.

Along branch Q2, representing systems one and three, there is much greater ambiguity. There are $2^3=8$ possibilities, indicated by **T₁** through **T₈** along the testing branch T. These are identified in the key to the decision tree in Annex I, and the damage associated with each outcome is recorded at the end of the branches. In half the outcomes, the preferred decision would be to regulate: in half, it would be not to regulate. The expected losses from non-regulation under various outcomes range from zero to 220.

Qualitative analysis provides guidance into the ex ante probabilities to be assigned to these outcomes. In fact, two possibilities are ruled out, because through examination of the structure of feedback, if $\partial F^2(t) / \partial T^2$ is negative in system three, it must also be negative in system one. If it is positive in system three, it may still be positive or negative in system one with equal likelihood. In system three the ex ante probabilities from qualitative analysis, for ease of computation, are now taken as 2/3 that **F¹** will be harmed, and 1/3 that **F²** will

be harmed, based on the preponderance of pathways with positive and negative contributions. Then, Table 4 records the eight possible outcomes of quantitative testing, the associated damages if the non-regulatory option is taken, and the probability of each outcome derived from qualitative analysis. Table 4 shows that qualitative analysis, by eliminating any likelihood of two outcomes for which the costs of the decisions not to regulate and not regulate do not differ by more than the costs of information, actually increases the value of further quantitative testing. More is at stake in choosing the correct option, so that the value of information which would eliminate the possibility of error is higher. So, the expected cost if quantitative testing is pursued under branch Q2 is 60 prior to testing costs and 70 thereafter, which compares favorably with the expected cost of 80 if the decision is taken based on the qualitative data alone.

The relevant expected costs when the possibility of further quantitative testing is considered are thus 65 along Q1, where such testing would not be pursued, and 70 along Q2, where it would be carried out, for an overall ex ante expected cost of $(1/2)(70+65)=67.5$. Not surprisingly, the value of qualitative information is higher, when there is a possibility of deciding subsequently to undertake further quantitative data collection prior to regulation. The benefit:cost ratio of qualitative information rises to 3.25:1. The use of qualitative analysis is complementary to complete quantitative analysis of relevant ecosystems, and fits naturally into a tiered testing system.

Table 4. Possible Outcomes, Associated Damages, and Probabilities Assigned on the Basis of Qualitative Analysis of Branch Q2

	System One	System Three		Damage
	F^2	F^1	F^2	
$T_1:$	$+(\frac{1}{2})$	$+(\frac{1}{3})$	$+(\frac{2}{3})$	$0(\frac{1}{9})$
$T_2:$	$+(\frac{1}{2})$	$-(\frac{2}{3})$	$+(\frac{2}{3})$	$20(\frac{2}{9})$
$T_3:$	$-(\frac{1}{2})$	$+(\frac{1}{3})$	$+(\frac{2}{3})$	$90(\frac{1}{9})$
$T_4:$	$-(\frac{1}{2})$	$-(\frac{2}{3})$	$+(\frac{2}{3})$	$110(\frac{2}{9})$
$T_5:$	$+(0)$	$+(\frac{1}{3})$	$-(\frac{1}{3})$	$90(0)$
$T_6:$	$+(0)$	$-(\frac{2}{3})$	$-(\frac{1}{3})$	$110(0)$
$T_7:$	$-(1)$	$+(\frac{1}{3})$	$-(\frac{1}{3})$	$180(\frac{1}{9})$
$T_8:$	$-(1)$	$-(\frac{2}{3})$	$-(\frac{1}{3})$	$220(\frac{2}{9})$

IV. CONCLUSION

The analysis has generated a comprehensive evaluation of testing alternatives in which expected benefits are compared with costs; also, a comparison of preferred regulatory options based on no information, qualitative information, and complete information about ecosystem functioning. Although the system and numbers used to demonstrate the methodology are hypothetical and considerably simplified, the exercise does lead to certain insights:

1. Regulatory decisions based solely on lab toxicity testing of individual species, without any consideration of species interactions, can lead to error and considerable excess loss;
2. Relatively simple qualitative analysis of ecosystem structure can lead to marked revision of prior probabilities based on lab testing; consequently, the value of this information can be high;
3. Prior qualitative analysis can provide considerable guidance as to the kinds of quantitative information about species interactions which would be helpful for regulatory decision making, or can obviate the need for such data;
4. It is possible to make reasonable judgements about the value of additional information, but these judgements necessarily depend, explicitly or implicitly, on judgements regarding the costs which would result from possible regulatory decisions in various states of the world; preliminary and approximate estimates of these costs are better than none; such esti

mates also inevitably enter into judgements about the extent of risk which constitutes "unreasonable adverse effects on the environment";

5. The techniques of qualitative analysis of ecological systems fit naturally into a decision framework designed to deal with uncertainty in regulatory policymaking towards testing and chemical risks to the environment; decision analysis enhances the usefulness of these techniques, which, in turn, are powerful aids in the assignment of probabilities in risk assessment.
6. Further research should be undertaken to apply these techniques to progressively more realistic regulatory problems.

APPENDIX

1. "A loop of length k is a simple, closed path from a variable to itself through k steps which visits each variable on the loop only once. The value of a loop is the product of the a_{ij} of its links, and the sign is the sign of that product. A loop of length 0 is by convention positive and has the value +1. Feedback is defined as the effect of a variable on itself by way of intervening variables.
2. Mathematically, the feedback at level k , (F_k) , in a system of $n \geq k$ variables is defined by $F_k = \sum (-1)^{m+1} L(m, k)$. Feedback at level k is summed over all sets of the products of m disjunct loops that total k elements. Disjunct loops have no variables in common (L =loops).
3. Loops of length 0 have a value of +1 and $F_0 = -1$. This is an algebraic convenience.
4. A path $P_{ij}^{(k)}$ is a product of $(k-1)$ alpha values from X_j to X_i involving k variables, none of which are visited more than once. $P_{ii} = 1$.
5. The complement of a path is the set of variables not on the path.
6. Let C_h be any of s parameters of the system $dx_i/dt = f_i(x_1, x_2, x_3, \dots; C_1, C_2, C_3, \dots, C_5)$. Then the effect of a change in C_n on the equilibrium level of any variable (\hat{X}_j) in the system is

$$\hat{X}_j / \partial C_h = \sum (\partial f_i / \partial C_h) \times P_{ji}^{(k)} \times F_{n-k} \{ \text{Comp } P_{ji}^{(k)} \} / F_n$$

that is, if C_n is a positive input to X_i ; then its effect on X_j will have the sign of the sum of the products of each path from X_i to X_j , each multiplied by the feedback of its complement, and all divided by the feedback of the whole. The input along that path has no effect. For this paper we postulate that the systems are stable.

Several qualitative results follow --

1. Since $F_n < 0$, if the complementary subsystem of a path is stable, its feedback is also negative, and $\partial \hat{X}_j / C_n$ has the same sign as the path products if they are all the same.
2. If the complement has zero feedback for all paths, the \hat{X}_j is independent of C_n .
3. If the complement of a path has positive feedback, the path has an effect of opposite sign to its own product.
4. The closer F_n gets to 0 (instability due to positive feedback equaling negative) the more sensitive all equilibrium values are to parameter change."

NOTES

1. 7U.S. Code 136a (C) (5-6) & (bb).
2. 40 CFR Part 158 Subpart E.
3. EPA, "Pesticides Registration; Proposed Data Requirements", 47FR53207, November 24, 1982.
4. National Research Council, Testing for Effects of Chemicals on Ecosystems, National Academy of Sciences, Washington, D.C., 1981.
5. National Research Council, op. cit.; p. 6.
6. A basic text is R.D. Luce and H. Raiffa, Games and Decisions. Introduction and Critical Survey, Wiley, 1957; an attempted application to problems of pesticide regulation is Mitre Corp., "The Application of Decision Theory to Toxicological Testing Requirements under FIFRA", prepared for Office of Pesticides Programs, U.S. EPA, Washington, D.C., 1977.
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12. Charles J. Puccia, "Qualitative models for the east coast benthos", 3rd Int. Conf. Ecol. Model., 1982.
13. May, op. cit.
14. Levins, op. cit.

15. Lane & Levins, op. cit.
16. It would be possible to reverse the order and investigate the systems quantitatively before deciding whether or not to ascertain their relative frequencies, and this alternative sequence could be analyzed, but is not, for simplicity's sake.
17. Support for this paper was received from EPA Grant CR 807809. The contents do not necessarily reflect the views or policies of EPA.

